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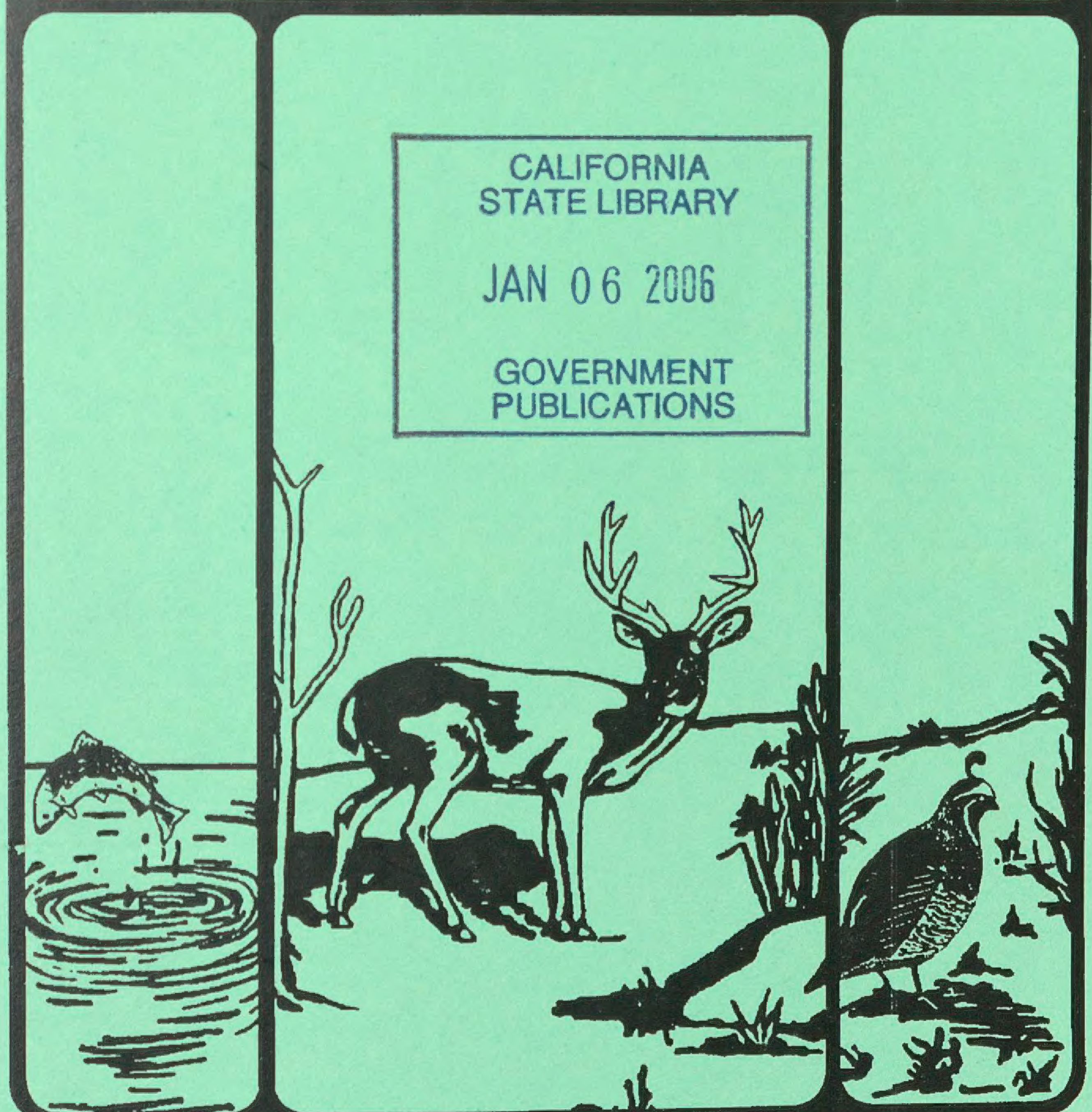

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Please direct correspondence to:

Ken Hashagen
Editor-in-Chief
California Fish and Game
P.O. Box 944209
Sacramento, California 94244-2090
e-mail: khashagen@mindspring.com

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RESPONSES OF FOREST SQUIRRELS TO GROUP-SELECTION TIMBER HARVESTING IN THE CENTRAL SIERRA NEVADA

BARRETT A. GARRISON
California Department of Fish and Game
Sacramento Valley-Central Sierra Region
1701 Nimbus Road
Rancho Cordova, CA 95670
e-mail: bgarrison@dfg.ca.gov

ROBIN L. WACHS
6198 Tule Lane
Foresthill, CA 95631
e-mail: rwachs@hotmail.com

MATTHEW L. TRIGGS
U.S. Forest Service, Tahoe National Forest
Foresthill Ranger District
22380 Foresthill Road
Foresthill, CA 95631
e-mail: mtriggs@fs.fed.us

Responses of Douglas' squirrels, *Tamiasciurus douglasii,* and western gray squirrels, *Sciurus griseus,* to group-selection timber harvesting were studied between 1995-2002 using point counts at four 21.2-ha stands in the central Sierra Nevada, California. Each stand had paired and adjacent 10.6-ha treatment and control plots, and four 0.6-ha group-selection harvest units were randomly placed and harvested in each treatment plot in 1998 and 1999. Effects were assessed using the Before-After/Control-Impact - Pairs (BACIP) design. The harvest reduced ($P < 0.05$) tree basal area (m^2/ha) and density (#/ha) by approximately 30% and snag density by approximately 45% within treatment stands. Douglas' squirrel populations differed among stands ($P < 0.001$) and among treatment and control plots nested within stands ($P < 0.079$). Douglas' squirrel populations were variously affected in the treatment plots during the post-treatment period because there was interaction ($P < 0.065$) between the stands and treatment period. Between-plot differences in Douglas' squirrel abundance varied among stands ($P = 0.002$) and there were greater post-treatment population declines ($P = 0.095$) in the control plots than treatment plots. These changes for Douglas' squirrels were not biologically significant because 90% confidence intervals overlapped for abundances and between-plot differences even though the mean difference from the post-treatment period differed by > 100% from the pre-treatment period. Populations of western gray squirrels did not vary ($P > 0.229$) among stands, plots, or treatment periods. There were equivalent between-plot differences for western gray squirrels between treatment periods ($P > 0.142$). There were no significant relationships with acorn crops for either squirrel for any population measure ($P > 0.340$). We concluded that the group-selection

harvesting had generally neutral effects to Douglas' squirrels and western gray squirrels due to the relatively small scale of ground and forest stand disturbance, relatively large amounts of unharvested adjacent forest, and retention of habitat attributes including oaks, *Quercus* spp., and pines, *Pinus* spp. Populations of these squirrels should be maintained in comparable habitat conditions when similarly implemented group-selection harvesting is done in the central Sierra Nevada.

INTRODUCTION

Conserving and managing forest resources of California's Sierra Nevada has been of considerable interest as demonstrated by two recent major resource assessment efforts (Centers for Water and Wildland Resources[1] 1996, U.S. Forest Service[2] 2001). These assessments concluded that timber harvesting was variously affecting the region's forested wildlife habitats, so harvesting impacts to wildlife are important considerations for forest and wildlife managers. Whether harvesting occurs on public or private lands, guidelines and regulations exist to ensure that wildlife needs are considered when logging operations are planned and conducted. Rarely, however, are these guidelines and regulations evaluated with field investigations of wildlife responses. Forest squirrels, such as Douglas' squirrels, *Tamiasciurus douglasii*, and western gray squirrels, *Sciurus griseus*, are conspicuous residents of forest habitats and economically important because they are common game species. These squirrels also have important ecological functions as consumers and dispersers of fungi, nuts from conifers and oaks, *Quercus* spp., and propagules of lichens and mosses (Carraway and Verts 1994, Steele 1999) so timber harvesting impacts to these species should be considered.

Douglas' squirrels are important indicators of overall ecosystem integrity of conifer-dominated forests in the Pacific States (Carey et al.[3] 1991, Carey 2000). The western gray squirrel occurs in forests and woodlands with oaks as dominant or codominant trees (Zeiner et al.[4] 1990, Carraway and Verts 1994), and they are indicators of ecosystem integrity (Ryan and Carey[5] 1995). Their populations are also particularly

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[1] Centers for Water and Wildland Resources. 1996. Sierra Nevada Ecosystem Project, final report to Congress, Volume I, assessment summaries and management strategies. University of California, Centers for Water and Wildlife Resources, Report No. 36, Davis, California, USA.

[2] U.S. Forest Service. 2001. Sierra Nevada Forest Plan Amendment, final environmental impact statement, Volume 1, Chapters 1 and 2. U.S. Department of Agriculture, Forest Service, Pacific Southwest Region, Vallejo, California, USA.

[3] Carey, A. B., B. L. Biswell, and J. W. Witt. 1991. Methods for measuring populations of arboreal rodents. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, Oregon. General Technical Report PNW-GTR-273.

[4] Zeiner, D. C., W. F. Laudenslayer, Jr., K. E. Mayer, and M. White, editors. 1990. California's wildlife Volume III mammals. State of California, The Resources Agency, Department of Fish and Game, Sacramento, California, USA. 407 pp.

[5] Ryan, L. A. and A. B. Carey. 1995. Biology and management of the western gray squirrel and Oregon white oak woodlands: with emphasis on the Puget Trough. U.S. Department of Agriculture, Forest Service, Pacific Northwest Research Station, Portland, Oregon. General Technical Report PNW-GTR-348.

responsive to the distribution and abundance of habitat attributes of older forests and woodlands such as downed woody debris, snags, trees with decadence, fungi, and nut and acorn producing trees (Buchanan et al. 1990, Carraway and Verts 1994, Steele 1999, Carey 2000, Pyare and Longland 2001). In addition, forest squirrels are important prey items for special-status predators such as the spotted owl, *Strix occidentalis*, (Gutierrez et al. 1995), northern goshawk, *Accipiter gentilis*, (Squires and Reynolds 1997, Watson et al. 1998), and fisher, *Martes pennanti*, (Zielinski et al. 1999). Knowledge of the effects of various forest harvesting options, including group-selection harvesting, to these squirrels is necessary if wildlife and forest managers are to consider these effects and adjust, if necessary, harvesting operations to meet current and future habitat needs of these squirrels.

Because it is thought to minimize impacts to wildlife, group-selection harvesting, where trees are harvested from small areas (< 1 ha), has been recommended as a silvicultural technique in the Sierra Nevada (McDonald and Tappenier[6] 1996). Group-selection harvesting also enhances site conditions for germination, sprouting, and growth for hardwoods and other tree species favoring openings in the forest canopy (McDonald and Tappenier[6] 1996). Group-selection is a regulated silvicultural technique for private and state lands in California (California Department of Forestry and Fire Protection[7] 2003). No studies to date, however, have been conducted on impacts to forest squirrels from group-selection harvesting in Sierra Nevada habitats where hardwoods such as California black oak, *Q. kelloggii*, are major components of these forests.

To evaluate effects of group-selection harvesting to forest squirrels in California black oak-dominated forest habitats in the Sierra Nevada, we used point counts to estimate squirrel populations and quantify their responses to the harvesting. Our null hypothesis was that populations of these two species of squirrels would not change with group-selection harvesting. With paired treatment and control plots in four replicated stands in the central Sierra Nevada, we measured responses by Douglas' and western gray squirrels by determining if differences occurred in squirrel abundances in the treatment and control plots between pre-treatment and post-treatment periods.

STUDY AREA

We conducted this study from 1995-2002 on four 21.2-ha study stands in southern Placer County, California. The term "stand" in this paper refers to the delineated study areas at the four sites but it still meets the ecological definition of a stand (Nyland 2002).

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[6] McDonald, P. M. and J. C. Tappeiner. 1996. Silviculture-ecology of forest-zone hardwoods in the Sierra Nevada. Pages 621-636 *in:* Sierra Nevada Ecosystem Project, final report to Congress, Volume III, assessments, commissioned reports, and background information. University of California, Centers for Water and Wildlife Resources, Report No. 38, Davis, California, USA.

[7] California Department of Forestry and Fire Protection. 2003. California Forest Practice Rules 2003. California Department of Forestry and Fire Protection, Forest Practice Program, Sacramento, California, USA. 275 pp.

Stands were at 1,220-1,320 m elevation, aspects were primarily east and south, and slopes between 5-30%. The stands were located on plateaus above canyons of tributaries of the South and Middle Forks of the American River. Study stands had tree layers dominated by large diameter (> 51 cm diameter breast height [dbh]) California black oak and ponderosa pine, *Pinus ponderosa*. Other less dominant tree species included Douglas-fir, *Pseudotsuga menziesii*, white fir, *Abies concolor*, sugar pine, *P. lambertiana*, and incense cedar, *Calocedrus decurrens*. Pole-sized and sapling California black oak and ponderosa pine dominated the midcanopy. The shrub layer was generally sparse, and deerbrush, *Ceanothus integerrimus*, and seedling California black oak were the most common shrubs. The herbaceous layer was dominated by occasional dense patches of mountain misery, *Chamaebatia foliolosa*. Group-selection harvesting was conducted at two stands in late September - early October 1998 and at two stands in late May 1999. In each stand, one 10.6-ha plot was randomly selected for harvesting and the adjacent 10.6-ha plot became the unharvested control. Harvesting was done on four 0.6-ha circular harvest units in each treatment plot. The size of the harvest units met recommended sizes for group-selection units where the distance between the unit's center and outer boundary was equal to twice the mean height of the adjacent unharvested trees (Nyland 2002). The harvest units were randomly selected from fifteen 0.04-ha vegetation sampling points that were randomly placed on a 25-m x 25-m sampling grid in each 10.6-ha plot. The total area harvested was 2.4 ha, which was 23% of the ground area in the treatment plot. All trees > 20 cm dbh were cut in the harvest units except 1-3 California black oaks retained in each harvest unit for monitoring acorn production. Retention also included all downed logs and snags that did not represent safety hazards. Rubber-tired skidders were used to remove logs from the harvest units. The group-selection harvest was a commercial timber sale with the U.S. Forest Service.

METHODS

Populations of Douglas' and western gray squirrels were sampled between July 1995 and June 2002 by direct visual or aural observations of individual animals using point counts. Point counts are often used to census and inventory landbirds (Ralph et al.[8] 1993). However, Carey et al.[3] (1991) used direct observations at counting points while Buchanan et al. (1990) used line transects to enumerate Douglas' squirrels. Hein (1997) also used line transects for counts of eastern gray squirrels, *Sciurus carolinensis*.

Studies of logging impacts to landbirds are generally conducted using counts of birds where species composition and populations are compared (Sallabanks et al. 2000).

[8] Ralph, C. J., G. R. Geupel, P. Pyle, T. E. Martin, and D. F. DeSante. 1993. Handbook of field methods for monitoring landbirds. U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station, Albany, California. General Technical Report GTR-PSW-144.

Landbird monitoring protocols developed by Ralph et al.[8 and 9] 1993 and 1995) were the foundation of the field methodology because landbirds were included in our impact assessment (Garrison et al. *In press*). Five permanent points were placed in each 10.6-ha plot yielding ten point count stations for each 21.2-ha stand. The five point-count stations in each plot were approximately 85-125 m apart and spaced in an "X" shape to uniformly survey the plot with the stations at the ends and intersection of the two intersecting lines.

At all count stations, one point count was done in each of the six 10-day periods between 1 May and 30 June, for a total of six counts. One count was also conducted each month between July and April. For data similarity, the three counts done each in May and June were averaged within the respective month to calculate a single monthly mean to be comparable with the monthly counts between July-April. Both squirrels are active the entire year and breeding activities begin in early winter and continue into spring and summer (Zeiner et al.[4] 1990, Carraway and Verts 1994, Steele 1999), so annual periods for analysis were September to August yielding 12 monthly counts in each annual period. September is the month of peak availability of acorns from oaks and nuts from pines and when squirrels were feeding upon and gathering acorns and nuts and hence more visible than at other times so it was appropriate to start the annual period then.

Point counts were conducted for 10 minutes at each station (Ralph et al.[8] 1993). All squirrels seen or heard during the 10-minute period were identified and tallied by the observers. Distances of each individual from the stations were estimated and categorized into ≤50-m and > 50-m belts. The maximum distances for detections were the outer boundaries of the 10.6-ha plots. Distance belts at each point were delineated with flagging, and observers calibrated their distance estimates repeatedly throughout the study. Duplicate observations of squirrels from previously counted points during the same day within the same plot or outside the plot boundaries were not included in the analysis. Point counts at each stand began at local sunrise and were completed in 3-3.5 hours.

Six observers counted squirrels over the 7-year study period but only 2-3 observers did counts in a given year. We followed the recommendations of Ralph et al.[9] (1995) to minimize observer bias. All observers were trained on point count methods, squirrel identification by sight and sound, and data recording. Each observer had a professional hearing screening, and all observers had hearing with normal levels. Observers were rotated among the four stands so that the number of observer visits was approximately equal for all stands each year. During each count day, the paired treatment and control plots at each stand were sampled so that temporal variation between plots was minimized. The order in which paired plots were counted was rotated on subsequent

[9]Ralph, C. J., S. Droege, and J. R. Sauer. 1995. Managing and monitoring birds using point counts: standards and applications. In: Monitoring bird populations by point counts (C. J. Ralph, J. R. Sauer, and S. Droege, eds.), pp. 161-175. U.S. Department of Agriculture, Forest Service, Pacific Southwest Research Station, Albany, California. General Technical Report PSW-GTR-149.

count days so that early and late morning surveys were evenly distributed between plots. Additionally, the order in which points were surveyed within plots was rotated so that the survey timing was evenly distributed among replicates within each plot.

Vegetative characteristics were sampled from fifteen 0.04-ha circular plots in each 10.6-ha plot. Circular plots were randomly selected and measured at intersection points of the 25-m by 25-m sampling grid in each 10.6-ha plot. Vegetation sampling was done July-August 1995 and August-September 1999 for the preharvest and postharvest measurements, respectively. Within the 0.04-ha plot, we measured diameters (cm) at 1.4 m height (dbh), heights (m), stem densities (number/ha), and basal areas (m^2/ha) of all live woody stems $\geq$13 cm dbh and $\geq$2 m tall. Shrub densities (# stems/ha) were determined from belt transects along the N-S and E-W transects where all live woody stems were counted that intersected a 1.0-m wide stick held 1.4 m above the ground. Snag (standing dead trees $\geq$13 cm dbh and $\geq$2 m tall) densities (number/ha) were enumerated from the same circular plots.

Acorn production was determined annually using visual counts of acorns (Garrison et al. 1998) from tagged trees in each stand. Each 10.6-ha subplot had 70-72 individually tagged trees > 13 cm dbh randomly located along transects. Some tagged trees were within the group-selection harvest units. To measure acorn production, a single observer counted all acorns observed during two 15-sec counts in randomly selected areas in the lower and upper halves of the tree's live crown. The two counts were totaled to yield a 30-sec count. Counts were done from late August to early October depending on the timing of each year's crop.

We used the BACIP (Before-After/Control-Impact - Pairs) experimental design (Stewart-Oaten et al. 1986 and 1992, Wiens and Parker 1995) where differences in squirrel abundances between the paired treatment and control plots in the pre-treatment and post-treatment periods were the primary response variable. General linear models (GLM) were used to test for significant differences in the between-plot abundance differences and squirrel abundances between the pre-treatment and post-treatment periods. General linear models are statistically equivalent to analysis of variance (ANOVA), and Underwood (1991, 1994) demonstrated that ANOVAs are the appropriate statistical analysis method for BACI designs with multiple treatment and control study locations. Because nesting and a covariate were used in our analysis (see below), GLMs were used instead of ANOVAs. The BACI design without paired (P) treatment and control plots was used to assess impacts from timber harvesting on Australian birds (Wardell-Johnson and Williams 2000), and Garrison et al. (2003, *In press*) used the BACIP design to assess group-selection harvesting impacts to breeding landbirds at these same study stands. Using differences between paired treatment and control plots during pre- and post-treatment periods is a conservative comparison (Wiens and Parker 1995) because it reduces the impact of the variability in animal counts with replicated counts and controls for natural differences between plots, time periods, and observers.

Two types of GLMs were used in the analysis. Two-way GLMs with annual acorn crops as a covariate were used to assess the statistical significance of between-plot abundance differences with stand and treatment period as fixed effects. Nested two-way GLMs with acorn crops as a covariate were used to assess the statistical

significance of differences in squirrel abundances between treatment periods, stands, and with plots nested within stands to account for plot effects. Acorn crop was used as a covariate because they are major dietary items for both squirrels, and it was the only measure we had available that could potentially account for other variation in squirrel populations that might be due to something other than the treatment. Plots were nested within stands because each pair of 10.6-ha treatment and control plots were unique to each study stand. Monthly between-plot differences and abundances were averaged across the 12 counts for each stand in each annual period, and these mean differences and abundances for the seven annual periods between 1995 and 2002 were response (dependent) variables in the GLMs. Significant results for treatment period, interactions between treatment period and stands, or interaction between treatment period and plots nested within stands represented significant treatment effects. We chose $\alpha < 0.10$ (instead of the traditional $\alpha < 0.05$) to test for significant differences to achieve balance between Type I and II errors (Steidl et al. 1997).

Transformations [$\log_{10}$(number counted + 1)] of the total number of individuals for each species at each plot for each replicate were done to improve the normal distribution of the data, and meet model assumptions of additivity and heteroscedasticity (Stewart-Oaten et al. 1986). Dependent variables were the differences between the $\log_{10}$ transformed abundances between plots and the mean annual counts for each plot nested within stands. Habitat changes due to the group-selection harvesting were assessed using t-tests of the mean differences between the pre- and post-treatment measurements for the paired plots. Confidence intervals (90%) were constructed for all squirrel data to confirm the statistical and biological significance of the statistical tests as recommended by Steidl et al. (1997), Johnson (2002), and Robinson and Wainer (2002). A difference of $\geq$ 100% between the pre- and post-treatment periods of the mean between-plot abundance differences was chosen as the threshold for a biologically significance difference following the recommendations of Steidl et al. (1997), a value used previously for impacts to birds with this study (Garrison et al. 2003, *In press*).

Pre-treatment and post-treatment periods were July-June 1995-1999 and May-June 1999-2002, respectively. Two stands were treated in September-October 1998 and all counts at these stands in 1998-1999 were done during the post-treatment period. Two stands were treated in May 1999 so counts from October 1998 to May 1999 were in the pre-treatment period while counts from June-September 1999 were in the post-treatment period. Each stand, therefore, had 3-4 years of complete or partial pre- and post-treatment data. All analyzes were conducted using SYSTAT statistical software (SPSS 2000a and 2000b).

RESULTS

Habitat Changes

Group-selection harvesting reduced ($P \leq 0.088$) densities (#/ha) and basal area (m^2/ha) of all trees (including conifers and hardwoods individually) by approximately 30% on the treatment plots (Table 1). Snag densities (#/ha) were reduced ($P = 0.033$) by 45%

Table 1. Means and 90% confidence intervals (CI) of habitat attributes and results of two-group t-tests of differences in attributes from treatment and control plots with pre-treatment (1995) and post-treatment (1999) measurements at four 21.2-ha study stands in Placer County, California.

Attribute	Plot	Pre-treatment (N = 4) Ave.	Pre-treatment (N = 4) 90% CI	Post-treatment (N = 4) Ave.	Post-treatment (N = 4) 90% CI	Differences between pre- and post-treatment levels[a] Ave.	Differences between pre- and post-treatment levels[a] 90% CI	T-test P-values[b] (df = 6)
Trees/ha	T[c]	254	137 – 371	178	75 – 280	-76	-109 – -44	**0.023**
	C	252	185 – 319	249	195 – 302	-3	-25 – 18	
Conifer trees/ha	T	149	23 – 275	108	5 – 211	-41	-70 – -13	0.074
	C	131	30 – 233	124	41 – 207	-7	-32 – 18	
Hardwood trees/ha	T	104	62 – 146	67	27 – 106	-37	-55 – -20	**0.014**
	C	121	40 – 201	124	45 – 204	4	0 – 7	
Tree basal area (m^2/ha)	T	35.8	26.5 – 45.1	25.5	15.5 – 35.5	-10.3	-16.1 – -4.4	**0.007**
	C	39.3	33.9 – 44.7	41.8	34.2 – 49.4	2.5	-1.2 – 6.1	
Conifer basal area (m^2/ha)	T	15.2	9.0 – 21.4	10.4	3.8 – 17.0	-4.8	-9.7 – 0.0	0.088
	C	20.1	11.6 – 28.6	20.7	11.1 – 30.3	0.6	-3.3 – 4.5	
Hardwood basal area (m^2/ha)	T	20.6	16.0 – 25.2	14.8	10.3 – 19.2	-5.8	-7.6 – -4.0	**0.001**
	C	19.2	12.8 – 25.6	21.0	16.1 – 26.0	1.8	-0.4 – 4.1	
Tree diameter (cm)	T	41.1	32.6 – 49.6	40.7	29.7 – 51.8	-0.4	-4.7 – 4.0	0.193
	C	41.5	31.0 – 51.9	43.6	33.8 – 53.3	2.1	0.9 – 3.4	
Conifer diameter (cm)	T	37.4	26.1 – 48.6	33.9	22.3 – 45.5	-3.5	-11.5 – 4.6	0.337
	C	46.6	29.9 – 63.2	47.8	27.9 – 67.6	1.2	-7.0 – 9.4	
Hardwood diameter (cm)	T	51.9	37.7 – 66.1	52.7	40.9 – 64.4	0.8	-2.3 – 3.9	0.443
	C	46.5	34.5 – 58.6	48.7	38.1 – 59.3	2.1	-0.3 – 4.6	
Tree height (m)	T	18.3	15.0 – 21.6	17.7	15.1 – 20.3	-0.5	-3.2 – 2.1	0.556
	C	20.0	15.1 – 24.9	20.3	15.4 – 25.2	0.3	-0.7 – 1.3	
Conifer height (m)	T	20.1	14.1 – 26.2	19.4	13.5 – 25.4	-0.7	-6.8 – 5.4	0.514
	C	25.4	17.0 – 33.9	26.6	17.6 – 35.6	1.2	0.0 – 2.5	

Hardwood height (m)	T	18.2	16.5 – 19.9	19.7	15.7 – 23.6	1.5	-1.7 – 4.7	0.243
	C	17.9	13.8 – 21.9	17.1	14.8 – 19.4	-0.7	-2.6 – 1.1	
Snags/ha	T	12.8	0.0 – 25.6	7.0	0.8 – 13.2	-5.8	-12.9 – 1.4	**0.033**
	C	16.9	4.7 – 29.1	18.9	10.3 – 27.5	2.1	-1.9 - 6.1	
Shrubs/ha	T	1520	-113 3152	1614	-889 4117	94	-922 – 1110	0.255
	C	1569	594 2544	1912	1199 2625	343	43 · 643	

[a] Mean differences between habitat attributes from treatment (upper value) and control (lower value) plots in 1995 and 1999.
[b] Significant P-values have **bold** font (*P* · 0.05) and underlined font (*P* · 0.10) from two-group t-tests of differences in log_{10}-transformed values between paired treatment and control plots between 1995 and 1999.
[c] T treatment plots; C control plots.

on the treatment plots (Table 1). Diameter (cm) and height of trees, including conifers and hardwoods, and shrub densities (# stems/ha) did not change ($P \geq 0.193$) with the group-selection harvesting (Table 1).

Squirrel Populations

Douglas' squirrels were two-five times more abundant than western gray squirrels in the four stands based on point count observations (Table 2). Douglas' squirrel populations and between-plot differences had considerably greater variation (measured by standard errors and 90% confidence intervals) among stands and among years than western gray squirrel populations (Tables 2 and 3). Most between-plot differences were negative for both squirrels indicating that populations were greater in control stands (Table 3).

Populations of Douglas' squirrels differed among stands ($P < 0.001$) and between plots nested within stands ($P < 0.079$) while western gray squirrel populations were similar ($P \geq 0.237$) when counts were adjusted for mean annual acorn crops at the four stands (Table 4). There was significant statistical interaction ($P = 0.065$) between stands and treatment period for Douglas' squirrel populations indicating varying treatment effects (Table 4). During the post-treatment period, Douglas' squirrel populations declined by approximately equal proportions in the control (-44%) and treatment plots (-43%) at Stand 3 indicating no treatment effects there. In the control plot at Stand 1, Douglas' squirrels had proportionately greater post-treatment increases than the treatment plot, while at Stand 4 post-treatment populations were proportionately greater in the control plot than the treatment plot. At Stand 2, Douglas' squirrel populations increased 21% in the treatment plot compared to an 18% decline in the control plot (Table 2). The 90% confidence intervals, however, overlapped for all pre- and post-treatment period stand and plot comparisons for Douglas' squirrels except the control plot at Stand 1 (Table 2). There was no interaction ($P = 0.846$) between stands and treatment period for western gray squirrels although there was considerable range in the relative differences (-6% to 275%) between the pre- and post-treatment period counts from the treatment and control plots for the four stands (Table 2).

Mean acorn crops at the four stands were used as a covariate and populations of both squirrels did not vary with acorn crops ($P \geq 0.825$) nor was there any interaction between acorn crops and stands ($P \geq 0.524$) (Table 4). When adjusted for mean annual acorn crops at the four stands, between-plot negative differences between annual counts from the paired treatment and control plots were found for Douglas' squirrels among stands ($P < 0.002$) and treatment periods ($P < 0.095$) but not western gray squirrels ($P > 0.142$) (Table 4). No statistical interaction ($P > 0.163$) between stand and treatment periods was found for either species, indicating that responses by both species were relatively consistent among stands (Table 4). For both species, between-plot differences did not vary with mean acorn crop ($P > 0.340$), and there was no interaction between stands and acorn crops ($P > 0.221$) (Table 4).

Between-plot differences in the post-treatment period increased by 667% for Douglas' squirrels and decreased by 85% for western gray squirrels compared to pre-

Table 2. Mean (SE) and 90% confidence intervals (CI) of annual counts of Douglas' squirrels and western gray squirrels between pre-treatment (1995-1999) and post-treatment (1999-2002) periods from paired 10.6-ha treatment and control plots from four 21.2-ha study stands in Placer County, California.

		Douglas' squirrel				Western gray squirrel			
		Pre-treatment (n = 4)		Post-treatment (n = 4)		Pre-treatment (n = 4)		Post-treatment (n = 4)	
Stand	Plot	Mean (SE)	90% CI	Mean (SE)	90% CI	Mean (SE)	90% CI	Mean (SE)	90% CI
1	T[a]	1.27 (0.20)	0.80 – 1.74	1.57 (0.69)	-0.05 – 3.19	0.17 (0.09)	-0.05 – 0.38	0.58 (0.25)	-0.02 – 1.17
	C	1.17 (0.24)	0.61 – 1.74	2.81 (0.35)	1.98 – 3.64	0.51 (0.18)	0.10 – 0.92	0.51 (0.22)	-0.02 – 1.03
2	T	0.14 (0.08)	-0.05 – 0.33	0.17 (0.05)	0.05 – 0.30	0.04 (0.03)	-0.03 – 0.10	0.15 (0.07)	0.00 – 0.31
	C	0.17 (0.09)	-0.04 – 0.39	0.14 (0.05)	0.03 – 0.25	0.26 (0.07)	0.09 – 0.43	0.32 (0.14)	-0.01 – 0.65
3	T	2.33 (0.92)	0.16 – 4.50	1.33 (0.22)	0.80 – 1.85	0.43 (0.08)	0.25 – 0.61	0.61 (0.23)	0.07 – 1.15
	C	1.45 (0.70)	-0.20 – 3.10	0.81 (0.14)	0.48 – 1.14	0.33 (0.13)	0.04 – 0.63	0.51 (0.20)	0.05 – 0.97
4	T	1.93 (0.86)	-0.09 – 3.95	0.92 (0.14)	0.58 – 1.25	0.32 (0.09)	0.10 – 0.54	0.30 (0.05)	0.18 – 0.42
	C	3.05 (1.16)	0.33 – 5.78	2.07 (0.22)	1.57 – 2.58	0.39 (0.09)	0.17 – 0.61	0.38 (0.09)	0.17 – 0.59
Mean	T	0.89 (0.12)	0.60 – 1.18	0.96 (0.03)	0.88 – 1.03	0.18 (0.07)	0.02 – 0.34	0.37 (0.08)	0.19 – 0.56
	C	0.87 (0.25)	0.29 – 1.45	1.35 (0.10)	1.11 – 1.60	0.35 (0.10)	0.13 – 0.58	0.40 (0.08)	0.22 – 0.58

[a] T = treatment plots; C = control plots.

Table 3. Mean (SE) and 90% confidence intervals (CI) of between-plot differences[a] of counts of Douglas' squirrels and western gray squirrels between pre-treatment (1995-1999) and post-treatment (1999-2002) periods at four 21.2-ha study stands in Placer County, California.

		Douglas' squirrel		Western gray squirrel	
Study Year	Treatment	Mean (SE) (n = 4)	90% CI	Mean (SE) (n = 4)	90% CI
1995-1996	Pre	-0.22 (0.24)	-0.78 – 0.35	-0.19 (0.09)	-0.41 – 0.03
1996-1997	Pre	0.12 (0.37)	-0.75 – 0.99	-0.18 (0.10)	-0.41 – 0.06
1997-1998	Pre	-0.01 (0.53)	-1.25 – 1.23	-0.20 (0.13)	-0.52 – 0.11
1998-1999	Pre	-0.13 (0.72)	-1.82 – 1.56	0.04 (0.16)	-0.33 – 0.41
Mean		-0.06 (0.41)	-1.02 – 0.90	-0.13 (0.10)	-0.37 – 0.11
1998-1999	Post	-0.93 (1.01)	-3.30 – 1.43	0.11 (0.05)	-0.02 – 0.23
1999-2000	Post	-0.43 (0.43)	-1.43 – 0.58	-0.15 (0.17)	-0.55 – 0.26
2000-2001	Post	-0.63 (0.41)	-1.60 – 0.35	0.01 (0.11)	-0.25 – 0.27
2001-2002	Post	0.14 (0.34)	-0.65 – 0.94	-0.04 (0.08)	-0.22 – 0.13
Mean		-0.46 (0.44)	-1.49 – 0.57	-0.02 (0.06)	-0.17 – 0.13

[a] Differences = treatment values – control values.

treatment period differences (Table 3). Therefore, the decline for the Douglas' squirrel was considered biologically significant because the differences were > 100%. There was, however, overlap in the 90% confidence intervals between the pre- and post-treatment periods for differences for both species, indicating that differences were equally variable among stands and treatment periods (Table 3). Furthermore, the 90% confidence intervals for the mean differences from the post-treatment period included 0.0 where impacts are numerically neutral (Table 3).

When acorn crops were removed as a covariate from the GLM's, similar statistical results (see following paragraph) were found for both squirrels for counts and differences indicating that acorns had little effect on overall treatment effects. Mean acorn crops displayed considerable variation among stands and across study years, and acorn crops were absent in most years at all stands and plots (Figure 1). In addition, squirrel populations did not track acorn production at the stand level. For example, Stand 2 had the greatest acorn crops but lowest squirrel populations, and there was considerable among-stand variation in squirrel populations and acorn crops.

Without acorns as a covariate, Douglas' squirrel populations varied among stands ($F = 16.58$; df = 3, 48; $P < 0.001$) and there was significant interaction between treatment period and stand ($F = 2.84$; df = 3, 48; $P = 0.047$). There was no significant variation between treatment periods, plots nested within stands, or with an interaction term for treatment period ($F = 0.39$-1.91; df = 1-4, 48; $P > 0.125$). Western gray squirrel populations varied among stands ($F = 3.94$; df = 3, 48; $P = 0.014$) but there was no

Table 4. Results of general linear models with stand, plot (nesting factor), and treatment period as fixed effects and mean annual California black oak acorn crop as the covariate on the yearly mean number of squirrels counted and yearly mean count differences between 1995-2002 at four 21.2-ha study stands in Placer County, California. Each study stand had paired adjacent 10.6-ha treatment and control plots.

	Douglas' squirrel					Western gray squirrel				
Source	SS	df	F-ratio	P-value	R^2	SS	df	F-ratio	P-value	R^2
Counts (N = 64)										
Stand	0.51	3	8.16	0.001	0.61	0.02	3	1.35	0.269	0.32
Treatment Period	0.01	1	0.42	0.518		0.01	1	1.44	0.237	
Stand (Plot)	0.19	4	2.25	0.079		0.02	4	1.16	0.342	
Stand * Treatment Period	0.16	3	2.58	0.065		0.01	3	0.27	0.846	
Treatment Period * Stand (Plot)	0.06	4	0.70	0.596		0.01	4	0.37	0.832	
Mean Acorn Crop	0.00	1	0.05	0.825		0.00	1	0.01	0.976	
Mean Acorn Crop * Stand	0.05	3	0.76	0.524		0.01	3	0.46	0.713	
Error	0.93	44				0.17	44			
Differences (N = 32)										
Stand	0.29	3	6.95	0.002	0.62	0.02	3	2.03	0.142	0.43
Treatment Period	0.04	1	3.06	0.095		0.01	1	1.62	0.218	
Stand * Treatment Period	0.08	3	1.89	0.163		0.01	3	1.01	0.408	
Mean Acorn Crop	0.01	1	0.96	0.340		0.00	1	0.75	0.397	
Mean Acorn Crop * Stand	0.07	3	1.60	0.221		0.01	3	0.76	0.530	
Error	0.28	20				0.06	20			

significant variation for the interaction and nested terms in the GLM's for treatment period and plots ($F = 0.29$-1.51; $df = 1$-$4, 48$; $P > 0.225$). With between-plot differences, there was significance variation among stands for Douglas' squirrels ($F = 6.42$; $df = 3, 24$; $P = 0.002$) and western gray squirrels ($F = 2.45$; $df = 3, 24$; $P = 0.088$). There was no significant variation between treatment period or interaction between stands and treatment period for both species ($F = 0.96$-2.73; $df = 1$-$3, 24$; $P > 0.111$). The GLM's did not account for all the variation in the response variables as the R^2 for both GLM's was 52-59% and 30-32% for Douglas' and western gray squirrels, respectively.

DISCUSSION

Group-selection timber harvesting had neutral effects on western gray squirrels over the 7-year period at four study stands in montane-hardwood conifer habitat in the central Sierra Nevada mountains of California. Western gray squirrel populations were very consistent among plots and treatment periods indicating that the group-selection harvest used in this study was essentially a biologically neutral habitat perturbation. The small size of western gray squirrel populations in our study stands as detected by point counts contributed to finding a neutral response because their populations were relatively low and approached 0.0 and remained so during the post-treatment period.

Group-selection timber harvesting had variable effects on Douglas' squirrels as their populations increased in treatment plots at two stands while populations either increased by a proportionately greater amount or decreased in control plots at the same stands. At the other two stands, Douglas' squirrel populations decreased by approximately equivalent amounts in the control and treatment plots in the post-treatment period. Because of these variable responses, we concluded that the overall group-selection harvesting effects on Douglas' squirrels were neutral because all but one of the 90% confidence intervals for the stand-plot treatment comparisons overlapped for pre- and post-treatment counts and included 0.0, and between-plot differences from the paired treatment and control plots and *P*-values from the GLM's for interactions between treatment period and stands or treatment period alone were $0.09 > P < 0.16$. Even though the mean post-treatment between-plot differences were > 100%, the predetermined threshold for significant biological effects, the variation among stand-level changes and population increases at two stands diluted this mean difference.

Populations of both squirrels are affected by many factors including food crops and habitat conditions (Koford[10] 1979, Foster[11] 1992, Carey 2000) so variable or neutral responses to group-selection harvesting are expected. The relatively small scale of the group-selection harvest, adjacent unharvested forest habitat, and retention of unmerchantable trees and tagged oaks in the harvest units all certainly combined to

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[10] Koford, R. R. 1979. Behavior and ecology of a California population of *Tamiasciurus douglasii*. PH.D. Dissertation, University of California, Berkeley, California, USA.

[11] Foster, S. A. 1992. Studies of ecological factors that affect the population and distribution of the western gray squirrel in northcentral Oregon. PH.D. Dissertation, Portand State University, Portland, Oregon, USA.

maintain squirrel populations in these four stands after the harvesting removed one-quarter of the trees. Both squirrels are highly territorial (Carraway and Verts 1994, Steele 1999), and timber harvesting removes trees in their territories, thereby affecting their populations. The spatial arrangement of harvested and unharvested areas influenced the population responses by these squirrels. Furthermore, the 3-4 yr pre-and post-treatment monitoring periods were relatively brief and may not have been long enough to detect longer term impacts. The duration of the two monitoring periods, however, was longer than other studies of forestry impacts to squirrels (Waters and Zabel 1998, Ransome and Sullivan 2002). There was also considerable variation in squirrel counts among stands and years that contributed to the equivocal responses.

Group-selection harvesting is an uneven-aged timber harvesting practice that results in small groupings of similar aged and size trees in otherwise larger forest stands (Arvola[12] 1978, Nyland 2002), and uneven-aged practices are generally thought to better maintain wildlife habitat conditions than even-aged practices such as clearcutting, rehabilitation, and seedtree and shelterwood removals (Hunter 1990, Kohm and Franklin 1997). Retaining unharvested areas with mature trees is known to maintain populations of forest squirrels. Adjacent unharvested habitat and retention of habitat features in harvest units reduced the severity of logging impacts to southern flying squirrels, *Glaucomys volans,* in Arkansas (Taulman et al. 1998). Douglas' squirrel winter populations were greater in old-growth than younger Douglas-fir forests in Washington because older and larger trees produce more cones than smaller and younger trees (Buchanan et al. 1990). Yet, Ransome and Sullivan (2003) found greater breeding recruitment for Douglas' squirrels in second-growth than old-growth coastal coniferous forests in British Columbia. Douglas' squirrel populations were lower in shelterwood-harvested forests compared to old-growth forests in northeastern California (Waters and Zabel 1998), but thinning in coastal coniferous forests in British Columbia had neutral effects (Ransome and Sullivan 2002).

Nuts from pines and acorns from oaks are important diet items from both squirrels (Steinecker and Browning 1970, Carraway and Verts 1994, Steele 1999). We measured annual production of acorns from California black oak but not nuts from ponderosa pine and other conifers in the study stands. In addition, we did not measure abundance of fungi, another important food for these squirrels (Stienecker and Browning 1970, Carraway and Verts 1994, Steele 1999). Therefore, we do not know if acorn and cone production are synchronous or if acorn, cone, and fungi abundance was affected by the group-selection harvest in our study stands, hence we cannot entirely conclude that squirrel populations were maintained by habitat retention that ensured consistently available food sources. Waters et al. (1994) found, however, that fungi abundance and biomass was equivalent in unthinned and thinned stands in red fir, *A. magnifica,* and white fir forests in northeastern California.

There was no statistical relationship between annual acorn crop and populations of both squirrels, but their populations may have fluctuated with pine cone crops

[12] Arvola, T.F. 1978. California forestry handbook. State of California, The Resources Agency, Department of Forestry, Sacramento, California, USA. 233 pp.

thereby affecting our analysis of the impacts of group-selection harvesting. Western gray squirrels have a geographic distribution that includes lower elevations than Douglas' squirrels (Zeiner et al.[4] 1990, Carraway and Verts 1994, Steele 1999) where oaks dominate compared to conifers, so oaks probably are of greater or lesser importance to western gray squirrel diets and populations as nuts from conifers, depending on the relative abundances of oaks and conifers. The corollary is true for the Douglas' squirrel in California, which has a distribution that includes higher elevations and northern coastal areas were conifers dominate (Zeiner et al.[4] 1990).

We feel, therefore, that acorn crop is an appropriate covariate measuring variation in food resources for populations of Douglas' and western gray squirrels in our study area. Using acorn crops also allowed us to account for an ecological factor affecting squirrel populations other than simply assessing habitat changes due to group-selection harvesting. In addition, the BACIP design accounts for inherent differences between study sites and unmeasured factors by analyzing pre- and post-treatment differences in simultaneous measures of response variables (Stewart-Oaten et al. 1986 and 1992; Wiens and Parker 1995). Furthermore, the small size of the study stands (21.2 ha) and the co-occurrence of the paired control and treatment plots on the study stands means that squirrel populations at both plots responded to the same annual food resource levels.

Ponderosa pines have heavy seed crops every third or fourth year, and ponderosa pines consistently have light to heavy seed crops more frequently than other conifers found in the central Sierra Nevada (McDonald 1992). In addition, the likely varying pattern of oak acorn and conifer nut crops likely ensures that one or both of these seasonally important food sources were available to both squirrels in all stands throughout the entire study period. In addition, light acorn crops occurred during both the pre- and post-treatment periods (Figure 1) so that squirrel populations were responding to changes in food availability concomitantly with habitat changes due to the group-selection harvesting.

Maintenance of biological diversity in forested habitats of the central Sierra Nevada is a land management goal currently advocated for public and private forestlands in the region so forest and wildlife managers should plan harvest operations that retain important habitat attributes for all wildlife. Douglas' squirrels and western gray squirrels are important members of the wildlife communities in the central Sierra Nevada so retention of habitat attributes used by both species should be implemented with any forestry operation along with retention of attributes used by all wildlife. With our study of group-selection harvesting, the small scale of ground and forest stand disturbance and retention of relatively large amounts of unharvested adjacent forest with mature pines and oaks that provided food sources resulted in neutral impacts to these two squirrels. Timber harvesting operations planned for habitats and locations similar to our study stands should retain and recruit similar habitat conditions to maintain populations of Douglas' and western gray squirrels. Group-selection timber harvesting can be used with these recommended retentions to maintain populations of forest squirrels.

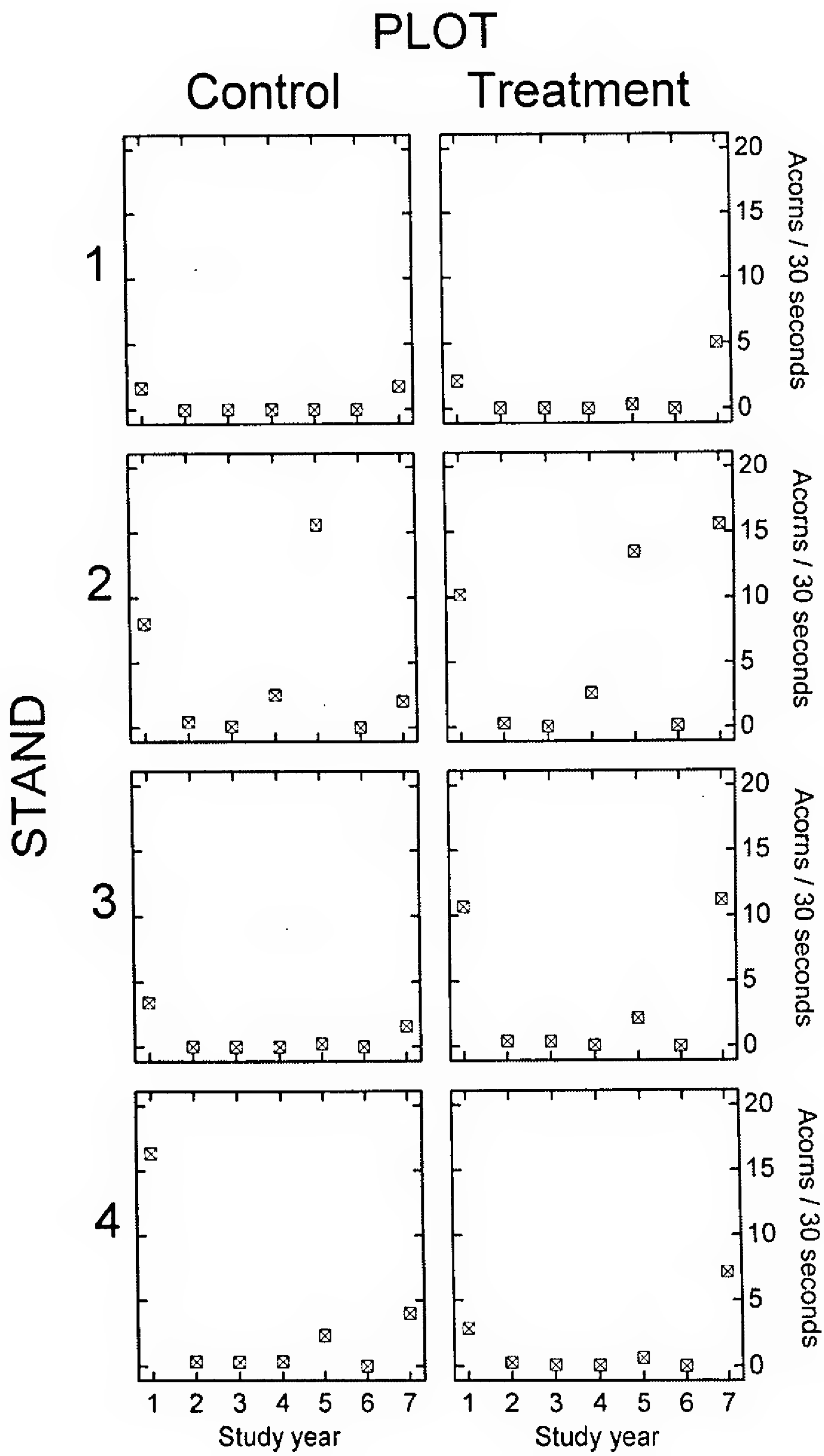

Figure 1. Mean annual acorn crops from 10.6-ha control and treatment plots between 1995 and 2002 from four 21.2-ha stands in Placer County, California. Study years 1-7 include acorn counts from 1995-2002, and mean counts were the number of acorns counted in a 30-sec period from 70-72 individually tagged California black oaks, *Quercus kelloggii*, at each plot. Group-selection harvesting occurred in Stands 3 and 4 between Study Years 3 and 4, and harvesting occurred in Stands 1 and 2 in Study Year 4.

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FISH BY-CATCH IN DUNGENESS CRAB, *CANCER MAGISTER*, RESEARCH TRAWLS OFF NORTHERN CALIFORNIA 1966-1969

Daniel W. Gotshall
Sea Challengers
4 Sommerset Rise
Monterey, CA 93940-4112
Gotsha6306@aol.com

John J. Geibel[1]
California Department of Fish and Game
284 Harbor Blvd.
Belmont, CA 94002
geibel@sbcglobal.net

Seventy-three species of fishes representing 27 families were captured in 227 research trawls off northern California from 1966 through 1969. The five most abundant fishes in 1966 and 1967 were Pacific tomcod, Pacific sanddab, rex sole, English sole, and night smelt. The five most frequently observed fishes over all 4 years were Pacific tomcod, Pacific sanddab, English sole, big skate, and butter sole. Twenty-six of these species are common in sport landings and 22 are of commercial value.

A number of these species have been declared over-fished and recent evidence indicates population declines in several additional species. Within the study area, commercial trawling for ocean shrimp, *Pandalus jordani*, and bottomfish occurs mostly in depths exceeding 55 m (30 fm).

INTRODUCTION

California Department of Fish and Game (DFG) marine biologists conducted trawling experiments off northern California in 1966, 1967, 1968, and 1969. These experiments were designed to determine the strength of the incoming year classes of Dungeness crabs, *Cancer magister*, (Gotshall 1978). Due to concern regarding the effect of fish by-catch in commercial trawls (Alverson 1996, University of Alaska 1996), we decided to analyze fishes caught in these crab trawls. This analysis was designed to determine the following:

1. Species composition of the fishes captured in the trawling areas.
2. Relative abundance of each species.
3. Bathymetric distribution of each species within the study area.
4. Species association.

This analysis documents relative species abundance prior to the rapid expansion of the trawl fishery and the increased use of roller gear that occurred in the 1970's and

[1]425 Central Ave, Menlo Park, CA 94025

1980's. Hopefully, this analysis can be used to assess changes over time in species composition, relative abundance, and depth distribution in this area.

METHODS

Trawling was conducted from DFG's research vessel N.B. Scofield in 1966 and 1968-1969. In 1967 DFG chartered the commercial trawler/crabber K.D.M. (Gotshall 1978). Trawling in 1966, 1967, and 1968 was conducted at randomly selected stations between False Cape (40°512N 124°392W) and Point St George (41°782N 124°262W) in depths of 18 m to 90 m (10 fm to 49 fm). In 1969, trawls were made along randomly selected east-west transects at approximately 9 m (5 fm) depth intervals from 18 m (10 fm) to 180 m (98 fm). The surveys were conducted in November and early December in 1966, in November during 1967 and 1968, and in October and November in 1969 (Willis[2] 1966, Taylor[3] 1967, Taylor[4] 1968, Nelson[5] 1969).

In 1966, trawls were made during daylight. In 1967 and 1968, trawls were conducted both during daylight and darkness and, in 1969, all trawls were made at night. We used a Gulf shrimp trawl with a 12.5 m (41 ft) head rope and 3.8 cm (1.4 inch) stretch mesh in 1966, 2.5 cm (1 inch) mesh in 1967, and 1.9 cm (0.75 inch) mesh in 1968 and 1969. The net was trawled for 0.8 km (0.5 mi) at each station (Gotshall 1978). Common and scientific names (Table 1) as well as the number of species in each family are from Robins et al. (1991). Depths of captures were grouped into 25 m (14 fm) bins.

Cluster analysis using S-PLUS version 6.1 from Insightful Corporation was run to analyze species associations in 1967 and 1969, the years with the widest range of trawl depths. The cluster analysis method was "agglomerative hierarchical clustering" using euclidean distance as the metric (S-PLUS 6 for Windows User's Guide, pp 427-30). Species with limited occurrence were excluded from the cluster analysis. Cluster analyses were run using occurrence by depth bin by year. Depth bins started at 25 m (14 fm) and extended to 150 m (82 fm) in 1967 and to 200 m (110 fm) in 1969 by 25 m (14 fm) intervals. The number of tows differed between years (Table 2). To minimize this affect both single and multiple occurrences in a depth bin were coded as "1".

RESULTS

Trawls were made at 227 stations over the 4 years of the study; 30 in 1966, 35 in 1967, 109 in 1968, and 53 in 1969 (Table 2). The incidental catch of fishes included 71 species representing 26 families (Table 1). The five most abundant fishes in 1966-1967 were Pacific tomcod, Pacific sanddab, rex sole, English sole, and night smelt (Table 3). The five most frequently observed species for all years combined were Pacific tomcod, Pacific sanddab, English sole, big skate, and butter sole (Tables 4, 5).

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[2]Willis, M. 1966. Cruise report 66-5-7 Crab.

[3]Taylor, S. N. 1967. Cruise report 67 Crab.

[4]Taylor, S. N. 1968. Cruise report 68-5.4 Crab.

[5]Nelson N. E. 1969. Cruise report 69-5.7 Crab.

Seven species occurred only once during the four surveys in the following years and depths: 1966: green sturgeon in 48 m (25 fm); 1967: king-of-the-salmon in 112 m (60 fm) and redtail surfperch in 20 m (10 fm); 1968: white sturgeon in 35 m (20 fm), plainfin midshipman in 94 m (50 fm), Pacific saury in 54 m (50 fm), and spotfin sculpin in 20 m (10 fm).

The 75-m (40-fm) depth bin produced the most species at 54 followed by the 50 m (27 fm) bin with 50, while the 175-m (96-fm) depth group produced the smallest number of species (16) (Figure 1).

Table 1. Family, scientific and common names of fish caught in crab tows

Family	Scientific Name	Common Name
Myxinidae	*Eptatretus stouti*	Hagfish, Pacific
Chimaeridae	*Hydrolagus colliei*	Ratfish, Spotted
Squalidae	*Squalus acanthias*	Dogfish, Spiny
Carcharhindae	*Mustelus henlei*	Smoothhound, Brown
Torpedinidae	*Torpedo californica*	Ray, Pacific Electric
Rajidae	*Bathyraja interrupta*	Skate, Sandpaper
	Raja binoculata	Skate, Big
	Raja rhina	Skate, Longnose
Clupeidae	*Alosa sapidissima*	Shad, American
	Clupea pallasi	Herring, Pacific
Engraulididae	*Engraulis mordax*	Anchovy, Northern
Acipenseridae	*Acipenser medirostris*	Sturgeon, Green
	Acipenser transmontanus	Sturgeon, White
Osmeridae	*Allosmerus elongatus*	Whitebait Smelt
	Hypomesus pretiosus	Smelt, Surf
	Spirinchus starksi	Smelt, Night
	Spirinchus thaleichthys	Smelt, Longfin
	Thaleichthys pacificus	Eulachon
Gadidae	*Merluccius productus*	Hake, Pacific
	Microgadus proximus	Tomcod, Pacific
Ophidiidae	*Chilara taylori*	Cusk-Eel, Spotted
Batrachoididae	*Porichthys notatus*	Midshipman, Plainfin
Scomberesocidae	*Cololabis saira*	Saury, Pacific
Trachipteridae	*Trachipterus altivelis*	King-Of-The-Salmon
Scorpaenidae	*Sebastes auriculatus*	Rockfish, Brown
	Sebastes crameri	Rockfish, Darkblotched
	Sebastes elongatus	Rockfish, Greenstriped
	Sebastes entomelas	Rockfish, Widow
	Sebastes flavidus	Rockfish, Yellowtail
	Sebastes goodei	Chilipepper
	Sebastes melanops	Rockfish, Black
	Sebastes paucispinis	Bocaccio
	Sebastes pinniger	Rockfish, Canary
	Sebastes babcocki	Rockfish, Redbanded
	Sebastes saxicola	Rockfish, Stripetail
	Sebastolobus alascanus	Thornyhead, Shortspine

Anoplopomatidae	*Anoplopoma fimbria*	Sablefish
Hexagrammidae	*Hexagrammos decagrammus*	Greenling, Kelp
	Ophiodon elongatus	Lingcod
Cottidae	*Artedius harringtoni*	Sculpin, Scalyhead
	Hemilepidotus hemilepidotus	Irish Lord, Red
	Hemilepidotus spinosus	Irish Lord, Brown
	Icelinus tenuis	Sculpin, Spotfin
	Leptocottus armatus	Sculpin, Staghorn
	Radulinus asprellus	Sculpin, Slim
Agonidae	*Agonopsis vulsa*	Spearnose, Northern
	Asterotheca infraspinata	Starnose, Spinycheek
	Occella verrucosa	Poacher, Warty
	Stellerina xyosterna	Poacher, Pricklebreast
	Xeneretmus latifrons	Poacher, Blackedge
Cyclopteridae	*Liparis fucensis*	Snailfish, Slipskin
	Liparis pulchellus	Snailfish, Showy
Embiotocidae	*Amphistichus rhodoterus*	Surfperch, Redtail
	Cymatogaster aggregata	Surfperch, Shiner
	Hyperprosopon ellipticum	Surfperch, Silver
	Phanerodon furcatus	Surfperch, White
Carangidae	*Trachurus symmetricus*	Mackerel, Jack
Zoarcidae	*Lycodes cortezianus*	Eelpout, Bigfin
	Lycodes pacifica	Eelpout, Blackbelly
Paralichyidae	*Citharichthys sordidus*	Sanddab, Pacific
Pleuronectidae	*Atheresthes stomias*	Flounder, Arrowtooth
	Lyopsetta exilis	Sole, Slender
	Lyopsetta jordani	Sole, Petrale
	Glyptocephalus zachirus	Sole, Rex
	Isopsetta isolepis	Sole, Butter
	Microstomus pacificus	Sole, Dover
	Parophrys vetulus	Sole, English
	Platichthys stellatus	Flounder, Starry
	Pleuronichthys coenosus	Sole, C-O
	Pleuronichthys decurrens	Sole, Curlfin
	Psettichthys melanostictus	Sole, Sand

Table 2. Number of Dungeness crab research trawl tows made by depth category by year. Depth category 25 contains depths of 25 m or less, 50 depths from 26 m to 50 m, etc.

YEAR	Depth (m)								Total
	25	50	75	100	125	150	175	200	
1966		23	7						30
1967	4	15	8	4	2	2			35
1968	7	41	49	12					109
1969	4	12	11	6	9	6	2	3	53
Total	15	91	75	22	11	8	2	3	227

Table 3. Total numbers of fish landed from crab trawls by species, depth category and year. Depth category is by 25 m intervals. Category 25 are depths less than 26 m, 50 are depths from 26 m to 50 m etc.

Common Name	Year · Depth							
	1966		1967					
	50	75	25	50	75	100	125	150
Smoothhound, brown	5	3		1	2			
Dogfish, spiny	16	46			1			14
Skate, big	88	19		10	3			
Ratfish, spotted						1		
Shad, american			1					
Herring, pacific		1	9	33		1		
Osmerid, unidentified			30					
Whitebait smelt	15	9	6	239	99	111	42	
Smelt, surf		1	4	77				
Smelt, night	115	116	148	362	35	46		
Eulachon	1				44	227	163	
Tomcod, pacific	6399	1970	93	1779	628	308	29	8
Hake, pacific		4			4	14	12	3
Sturgeon, green	1							
Anchovy, northern		20		1	2			
Snailfish, showy	752	23		13	1			
Eelpout, bigfin						1		
King-of-the-salmon							1	
Poacher, warty				3				
Poacher, pricklebreast	114	33	1	10				
Poacher, blackedge						1		
Surfperch, redtail			1					
Surfperch, shiner	38	43	8	40	329	6	1	
Surfperch, white	3			2				
Rockfish, darkblotched					2	35	32	2
Rockfish, greenstriped								1
Rockfish, yellowtail		1						4
Rockfish, black	152	17	3	11				
Rockfish, canary		1				3		1
Irish lord, brown		1						
Sculpin, staghorn	7	3	2					
Sculpin, scalyhead	5	2						
Mackerel, jack			2	13	2			
Greenling, kelp	1	1						
Lingcod	1	1						
Sablefish		1			66	51	20	29
Sanddab, pacific	107	298	1	150	319	239	70	4
Flounder, arrowtooth								2
Sole, petrale	1	5			3			1
Sole, rex		51			189	272	213	240
Sole, butter	298	88	3	93	2	1		
Sole, slender					15	26	50	95
Sole, dover		11			13	17	16	20
Sole, english	221	456		68	116	14	8	4
Flounder, starry	80		1	15				
Sole, c-o				2	1	2	1	
Sole, curlfin	1	1						
Sole, sand	35	5	3	6				

Table 4. Occurance per crab trawl by species, depth category and year for species occuring in multiple tows. Depth category is by 25 m intervals. Category 25 are depths less than 26 m, 50 are depths from 26 m to 50 m, etc.

Common Name	1966		1967						1968				1969							
	50	75	25	50	75	100	125	150	25	50	75	100	25	50	75	100	125	150	175	200
Hagfish, pacific											0.12	0.08			0.18	0.17	0.44			0.33
Smoothhound, brown	0.17	0.14		0.07	0.13				0.29	0.29	0.24		0.25	0.33	0.09					
Dogfish, spiny	0.17	0.57			0.13			0.50	0.14	0.02	0.06	0.08	0.25				0.11			
Ray, pacific electric											0.04				0.09		0.11			
Skate, sandpaper																	0.11	0.33	1.00	0.67
Skate, big	0.91	0.57		0.27	0.25				1.00	0.93	0.71	0.42	1.00	0.92	0.64	0.33	0.89	0.83	1.00	0.33
Skate, longnose									0.14	0.02	0.02			0.08	0.18	0.33	0.78	0.83	0.50	0.33
Ratfish, spotted						0.25				0.17	0.20	0.17			0.09		0.11	0.17		0.33
Herring, pacific		0.14	0.25	0.47		0.25				0.05	0.35	0.50		0.08	0.36	0.17				
Whitebait smelt	0.17	0.14	0.50	0.67	0.88	0.75	1.00		0.57	0.29	0.57	0.42	0.50	0.33	0.18		0.11			
Smelt, surf		0.14	0.50	0.13						0.05			0.25		0.09					
Smelt, longfin										0.02	0.06		0.25	0.17	0.09					
Smelt, night	0.61	1.00	0.75	0.73	0.63	0.75			0.57	0.46	0.39	0.17	0.50	0.50	0.27					
Eulachon	0.04				0.75	1.00	1.00				0.12	0.33			0.09	0.33	0.22			
Tomcod, pacific	1.00	1.00	0.75	1.00	1.00	1.00	0.50	1.00	1.00	0.98	0.96	1.00	1.00	1.00	0.91	0.83	0.44	0.17		
Hake, pacific		0.14			0.50	0.75	1.00	0.50		0.02	0.27	0.08		0.08	0.27	1.00	1.00	0.83	1.00	1.00
Anchovy, northern		0.14		0.07	0.25				0.29	0.17	0.10			0.08	0.18					
Snailfish, slipskin									0.14	0.10	0.04									
Snailfish, showy	0.74	0.86		0.33	0.13				0.57	0.41	0.51	0.58	0.50	1.00	0.82	0.33	0.11			
Eelpout, bigfin						0.25						0.08			0.09	0.67	0.56	0.50	1.00	1.00
Eelpout, blackbelly											0.04	0.08			0.09		0.33	0.17		
Cusk-eel, spotted																0.33	0.44	0.17		
Poacher, warty				0.20					0.29	0.27	0.10	0.17		0.42	0.27					
Poacher, pricklebreast	0.70	0.57	0.25	0.20					0.57	0.59	0.39		1.00	0.83	0.27		0.11			
Spearnose, northern											0.02			0.08	0.18					
Starnose, spinycheek												0.08		0.08		0.17	0.11			
Poacher, blackedge						0.25												0.33	1.00	1.00
Surfperch, shiner	0.39	0.71	0.75	0.60	0.88	0.25	0.50		0.43	0.37	0.67	0.58	0.25	0.58	0.73	0.50	0.11			
Surfperch, silver										0.05	0.02									
Surfperch, white	0.09			0.13						0.02										
Rockfish, brown											0.04									
Rockfish, darkblotched					0.13	1.00	1.00	1.00			0.04	0.67			0.09	0.67	0.78	0.67		
Rockfish, greenstriped								0.50									0.33	0.50	1.00	1.00

Rockfish, yellowtail		0.14						0.50			0.04	0.25					0.11	0.17		
Chilipepper											0.02					0.17			0.50	
Rockfish, black	0.61	0.43	0.50	0.40					0.71	0.54	0.31	0.08	1.00	0.42	0.09		0.11			
Rockfish, canary		0.14				0.25		0.50			0.02			0.08	0.09					
Rockfish, stripetail																		0.50	1.00	1.00
Thornyhead, shortspine																				1.00
Irish lord, brown		0.14											0.25	0.08						
Sculpin, staghorn	0.26	0.43	0.50						0.86	0.46	0.14		0.50	0.33	0.18					
Sculpin, slim																	0.33	0.67	0.50	1.00
Sculpin, scalyhead	0.09	0.14																		
Mackerel, jack			0.25	0.20	0.25															
Greenling, kelp	0.04	0.14																		
Lingcod	0.04	0.14								0.12	0.20	0.17	0.25	0.08	0.09					0.67
Sablefish		0.14			0.63	1.00	1.00	1.00		0.07	0.27	0.50		0.08	0.36	1.00	1.00	1.00	1.00	1.00
Sanddab, pacific	0.52	1.00	0.25	0.93	1.00	1.00	1.00	1.00	0.29	0.73	0.92	0.83	0.50	0.75	1.00	1.00	1.00	0.50		
Flounder, arrowtooth								0.50			0.06	0.42			0.18	0.83	0.78	0.33	1.00	
Sole, petrale	0.04	0.29			0.25			0.50	0.14	0.17	0.41	0.50	0.25	0.17	0.73	0.83	0.22	0.33		0.33
Sole, rex		0.43			0.88	1.00	1.00	1.00		0.15	0.41	0.92		0.17	0.64	1.00	1.00	0.83	1.00	0.67
Sole, butter	0.91	1.00	0.50	0.93	0.25	0.25			0.86	0.66	0.59	0.25	1.00	0.75	0.64					0.33
Sole, slender					0.63	1.00	1.00	1.00		0.05	0.14	0.58			0.36	0.83	1.00	1.00	1.00	1.00
Sole, dover		0.29			0.50	0.50	0.50	1.00		0.10	0.49	0.67		0.17	0.82	1.00	1.00	1.00	1.00	1.00
Sole, english	0.65	1.00		0.47	0.88	1.00	0.50	0.50	0.71	0.88	0.90	0.75	0.50	0.50	0.82	0.33	0.22	0.67		
Flounder, starry	0.52		0.25	0.33					0.29	0.71	0.10	0.25	0.50	0.42	0.09		0.11			
Sole, c-o				0.13	0.13	0.25	0.50			0.15	0.16	0.17								
Sole, curlfin	0.04	0.14								0.02	0.02				0.09					
Sole, sand	0.65	0.43	0.75	0.33					0.86	0.54	0.27	0.08	0.75	0.67	0.36					

Table 5. Occurance in crab trawls by species, depth category and year for species occuring in multiple tows. Depth category is by 25 m intervals. Category 25 are depths less than 26 m, 50 are depths from 26 m to 50 m, etc.

Common Name	Year / Depth																			
	1966		1967						1968				1969							
	50	75	25	50	75	100	125	150	25	50	75	100	25	50	75	100	125	150	175	200
Hagfish, pacific											6	1			2	1	4			1
Smoothhound, brown	4	1		1	1				2	12	12		1	4	1					
Dogfish, spiny	4	4			1			1	1	1	3	1	1				1			
Ray, pacific electric											2				1		1			
Skate, sandpaper																	1	2	2	2
Skate, big	21	4		4	2				7	38	35	5	4	11	7	2	8	5	2	1
Skate, longnose									1	1	1			1	2	2	7	5	1	1
Ratfish, spotted						1				7	10	2			1		1	1		1
Herring, pacific		1	1	7		1				2	17	6		1	4	1				
Whitebait smelt	4	1	2	10	7	3	2		4	12	28	5	2	4	2		1			
Smelt, surf		1	2	2						2			1		1					
Smelt, longfin										1	3		1	2	1					
Smelt, night	14	7	3	11	5	3			4	19	19	2	2	6	3					
Eulachon	1				6	4	2				6	4			1	2	2			
Tomcod, pacific	23	7	3	16	8	4	1	2	7	40	47	12	4	12	10	5	4	1		
Hake, pacific		1			4	3	2	1		1	13	1		1	3	6	9	5	2	3
Anchovy, northern		1		1	2				2	7	5			1	2					
Snailfish, slipskin									1	4	2									
Snailfish, showy	17	6		5	1				4	17	25	7	2	12	9	2	1			
Eelpout, bigfin						1						1			1	4	5	3	2	3
Eelpout, blackbelly											2	1			1		3	1		
Cusk-eel, spotted																2	4	1		
Poacher, warty				3					2	11	5	2		5	3					
Poacher, pricklebreast	16	4	1	3					4	24	19		4	10	3		1			
Spearnose, northern											1			1	2					
Starnose, spinycheek												1		1		1	1			

Poacher, blackedge						1												2	2	3
Surfperch, shiner	9	5	3	9	7	1	1		3	15	33	7	1	7	8	3	1			
Surfperch, silver										2	1									
Surfperch, white	2			2						1										
Rockfish, brown											2									
Rockfish, darkblotched					1	4	2	2			2	8			1	4	7	4		
Rockfish, greenstriped								1									3	3	2	3
Rockfish, yellowtail		1						1			2	3					1	1		
Chilipepper											1					1			1	
Rockfish, black	14	3	2	6					5	22	15	1	4	5	1		1			
Rockfish, canary		1				1		1			1			1	1					
Rockfish, stripetail																		3	2	3
Thornyhead, shortspine																				3
Irish lord, brown	1												1	1						
Sculpin, staghorn	6	3	2						6	19	7		2	4	2					
Sculpin, slim																	3	4	1	3
Sculpin, scalyhead	2	1																		
Mackerel, jack			1	3	2															
Greenling, kelp	1	1																		
Lingcod	1	1								5	10	2	1	1	1					2
Sablefish		1			5	4	2	2		3	13	6		1	4	6	9	6	2	3
Sanddab, pacific	12	7	1	14	8	4	2	2	2	30	45	10	2	9	11	6	9	3		
Flounder, arrowtooth								1			3	5			2	5	7	2	2	
Sole, petrale	1	2			2			1	1	7	20	6	1	2	8	5	2	2		1
Sole, rex		3			7	4	2	2		6	20	11		2	7	6	9	5	2	2
Sole, butter	21	7	2	14	2	1			6	27	29	3	4	9	7					1
Sole, slender					5	4	2	2		2	7	7			4	5	9	6	2	3
Sole, dover		2			4	2	1	2		4	24	8		2	9	6	9	6	2	3
Sole, english	15	7		7	7	4	1	1	5	36	44	9	2	6	9	2	2	4		
Flounder, starry	12		1	5					2	29	5	3	2	5	1		1			
Sole, c-o				2	1	1	1			6	8	2								
Sole, curlfin	1	1								1	1				1					
Sole, sand	15	3	3	5					6	22	13	1	3	8	4					

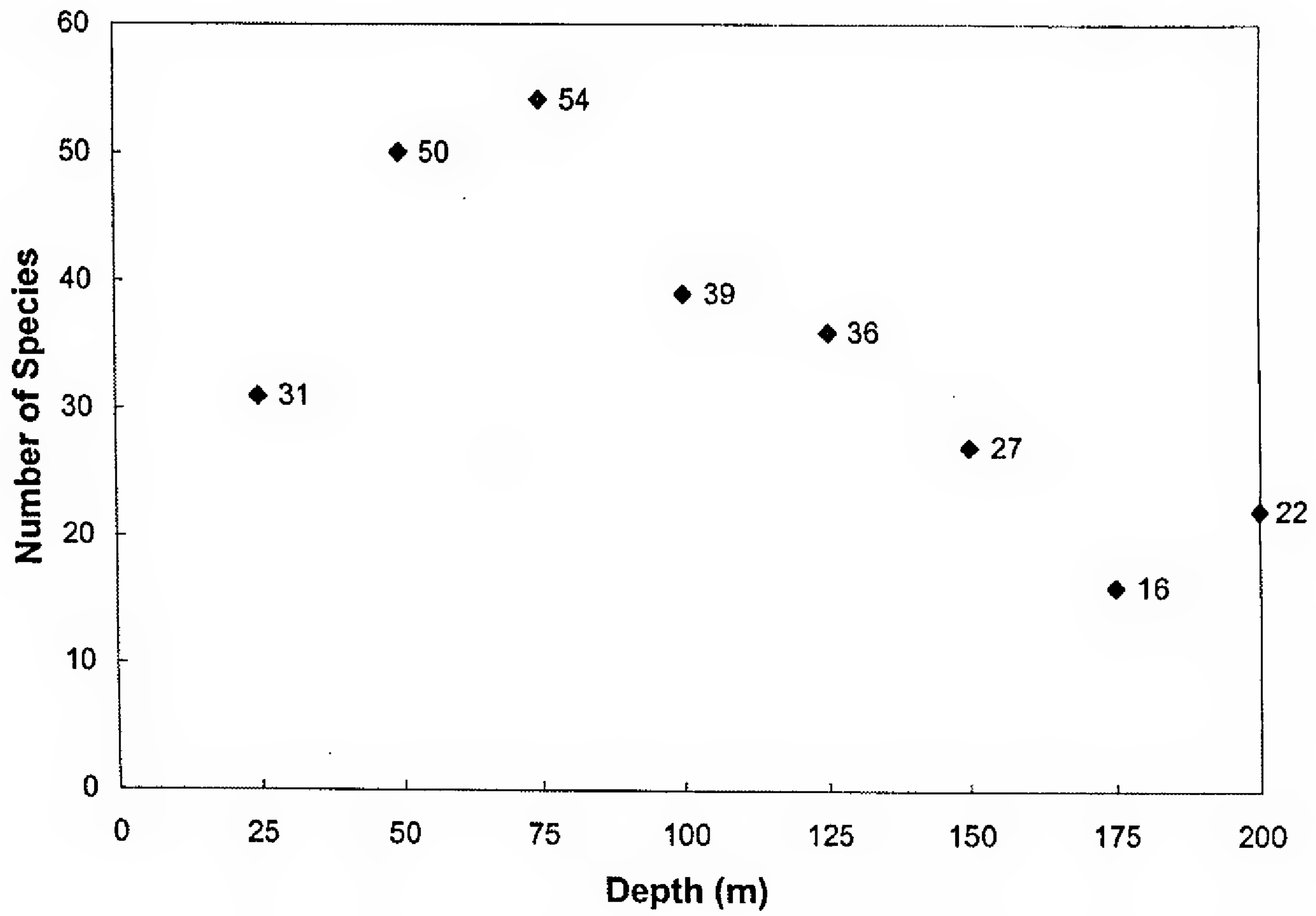

Figure 1. Number of species of fish captured in crab research trawls by depth (m) for all years of the survey, 1966-1969.

Results by Family

Family Myxinidae – Hagfish

Only one species of hagfish was encountered. The Pacific hagfish occurred in 6% of the trawls and in depths in excess of 50 m (27 fm) (Table 4). Since these scavengers can easily escape through even the small mesh used on the cod end, this number probably underestimates abundance.

Family Chimaeridae – Ratfish

The spotted ratfish was captured in 10% of the trawls and in depths greater than 25 m (15 fm) (Table 4). Based on our observations, spotted ratfish are more common at greater survey depths.

Family Carcharhinidae - Requiem Sharks

One species of requiem shark, the brown smoothhound, was captured in the crab trawls. A total of 11 of these small sharks was captured during 1966 and 1967 (Table 3). They were most frequently caught at a depth of 35 m (20 fm) (Table 4).

Family Squalidae - Dogfish Sharks

Spiny dogfish were captured from the shallowest trawls down to a depth of 125 m (70 fm) (Table 4).

Family Torpedinidae - Electric Rays

The Pacific electric ray, the only member of the family recorded from California, was captured in four tows, two in 1968 and two in 1969 (Table 5). Capture depths ranged from 50 m to 125 m (27 fm to 68 fm) (Table 4).

Family Rajidae – Skates

The big skate was the most frequently observed of the three species caught, occurring in over 60% of the trawls in all four years and at all depths during the study (Table 4). Longnose skate was caught in 1968 and 1969 from shallow to deep tows while sandpaper skate was only caught in 1969 (Table 4).

Family Clupeidae – Herrings

Our trawling captured two of the seven species of herring that occur off California. American shad was only caught on the shallowest tows, while Pacific herring was encountered more frequently and over a greater depth range (Table 4). A few of these pelagic fishes were probably captured in mid-water as nets were being deployed or retrieved.

Family Engraulidae – Anchovies

The northern anchovy (Table 4) was most likely captured in mid-water when nets were being deployed or retrieved.

Family Osmeridae – Smelts

Five of the seven species of marine smelts that occur in California waters were recorded. The most abundant in the 1966 and 1967 tows were night smelt, followed by whitebait smelt and eulachon (Table 3). None of these species was captured in waters deeper than 110 m (60 fm) (Table 4).

Family Gadidae – Cods

Two of the four species of cods that occur in California were taken. The Pacific tomcod was the most abundant and most frequently captured in the trawls (Table 4). These small cod were taken in all depths of the survey but were far more common in

the shallower trawls (Table 3). Pacific hake occurred more frequently in deeper trawls but were encountered in all but the shallowest trawls (Table 4).

Family Ophidiidae - Cusk-Eels

The spotted cusk-eel was captured only in 1969 trawls at depths ranging from 75 m (40 fm) to 150 m (80 fm) (Table 4).

Family Scorpaenidae – Scorpionfishes

Fifty-nine species of this large family have been reported off the California coast (Lea et al. 1999). In the four survey years, we captured 12 of these species. The most frequently observed was black rockfish, followed by the darkblotched rockfish (Table 3). Black rockfish was captured as deep as 125 m (70 fm), while darkblotched rockfish ranged in depths of capture from 50 m to 150 m (27 fm to 80 fm) (Table 4). Black rockfish, yellowtail rockfish, and canary rockfish were the only rockfish caught in every year of the survey (Table 4).

Family Anoplopomatidae – Sablefish

Sablefish, an important commercial species, was frequently captured in our trawls (Table 3). These fish are more common in depths beyond 75 m (40 fm) however we captured them in depths as shallow as 37 m (20 fm) (Table 4).

Family Hexagrammidae – Greenlings

This family is restricted to the temperate Pacific Ocean; two of the nine species that occur off the Pacific Coast were captured. Lingcod was encountered in 1966, 1967, and 1968, while kelp greenling was captured only in 1966 (Table 4). Lingcod captures occurred from the shallowest to the deepest trawls. Only two kelp greenling were captured one at 45 m (25 fm) and the other at 50 m (27 fm) (Table 3).

Family Cottidae – Sculpins

Six species of cottids were observed in catches. Staghorn sculpin, the most common sculpin encountered, occurred in 22% of the trawls. Depth of capture ranged from 20 m to 75 m (10 fm to 40 fm) (Table 5).

Family Agonidae – Poachers

The pricklebreast poacher was the most frequently captured of the five species of poachers occurring in 35% of the trawls. The pricklebreast poacher was caught at shallower depths (Table 5).

Family Cyclopteridae – Snailfishes

The showy snailfish was the most frequently observed of the two species of snailfish that were captured. Showy snailfish was most often caught in the 50-m depth group (27 fm), but were caught as deep as 125 m (70 fm). The slipskin snailfish was much rarer and caught in depths shallower than 75 m (40 fm) (Table 4).

Family Carangidae – Jacks

Jack mackerel are pelagic and only 17 were captured in six trawls all in 1967. These fish may have been caught during deployment or retrieval of the trawl gear (Table 3).

Family Embiotocidae – Surfperches

Four species of surfperches were captured during the trawling. Shiner surfperch were the most frequently observed. This small surfperch was captured as deep as 125 m (68 fm) and were most common in the 75 m (40 fm) depth group (Table 4).

Family Zoarcidae – Eelpouts

We captured only two species: bigfin eelpout and blackbelly eelpout. Bigfin eelpout was more common at the deeper stations in the survey, but we captured them at depths as shallow as 73 m (40 fm) (Table 4).

Family Paralichthyidae – Lefteye Flounders

The Pacific sanddab was the only lefteye flounder encountered during the study. They occurred in 75% of the trawls (Table 5). Pacific sanddab was captured as deep as 120 m (66 fm) and in all of the 90 m and 110 m (50 fm and 60 fm) group trawls.

Family Pleuronectidae - Righteye Flounders

Twenty-five members of this family occur off our coast, and we captured 11 of these. The most frequently encountered were English sole, butter sole, sand sole, and Dover sole, respectively. English sole were encountered out to 150 m (80 fm) and Dover sole, a deep-water species, out to 180 m (100 fm) (Table 4).

Cluster Analysis Results

Three assemblages were consistent between 1967 and 1969. Starry flounder, prickleback poacher, and black rockfish had a close association based on depths of captures. Pacific hake, sablefish, dover sole, and rex sole formed another close association. English sole, Pacific sanddab, and Pacific tomcod formed the third group. Other species formed cluster that differed between the 2 years (Figure 2 and Figure 3).

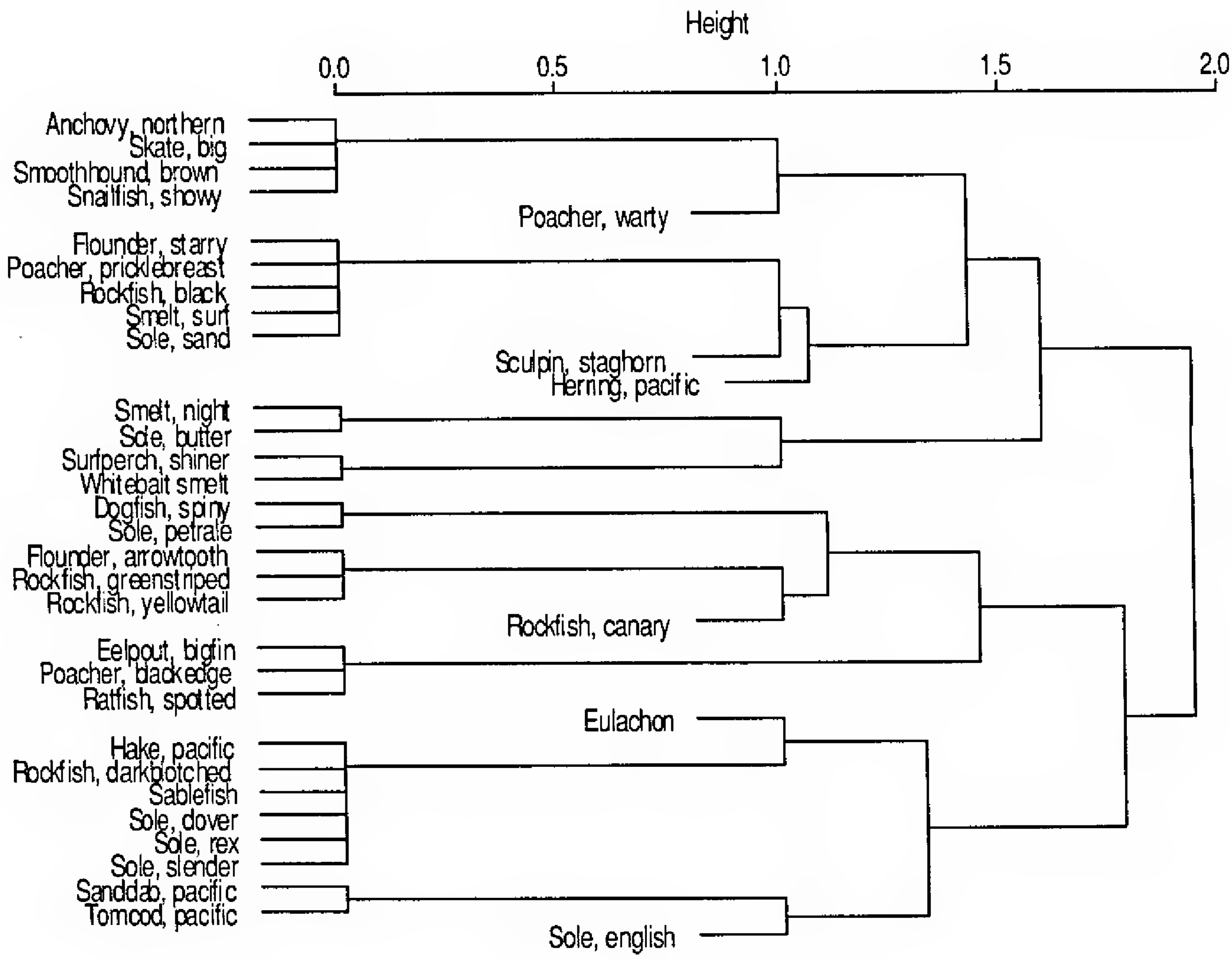

Figure 2. Cluster analysis using frequency of occurrence by depth for depth bins from 25m to 150m by 25m intervals for 1967.

DISCUSSION

Commercial trawlers are restricted to waters outside California's 3-mile limit in northern California (regulations allow shrimp trawlers to fish within 2.5 miles at several locations). Between Cape Mendocino and the California-Oregon border, trawlers target numerous species, including flounders, rockfishes, sablefish, lingcod, and ocean shrimp. Bottomfish trawl nets have a minimum legal mesh size of 116 mm (4.5 inch), while shrimp trawl nets have a minimum mesh size of 3.5 cm (1.4 inch).

Most of the area outside of state waters is located in depths in excess of 55 m (30 fm); however, there are a few areas where the 3-mile limit shallows to 37 m (20 fm). In this survey, 83% (61/73) of the species were captured in water 55 m (30 fm) or deeper (Table 4). Of these, 35 species are important to commercial and/or sport fisheries and 22 are targeted by commercial trawlers (Leet[6] 2002). Many of the other species were observed in shrimp trawls monitored in 1964 by the senior author.

[6]Leet W. S. 2002. California's living marine resources: A status report. University of California Pub. SG01-11.

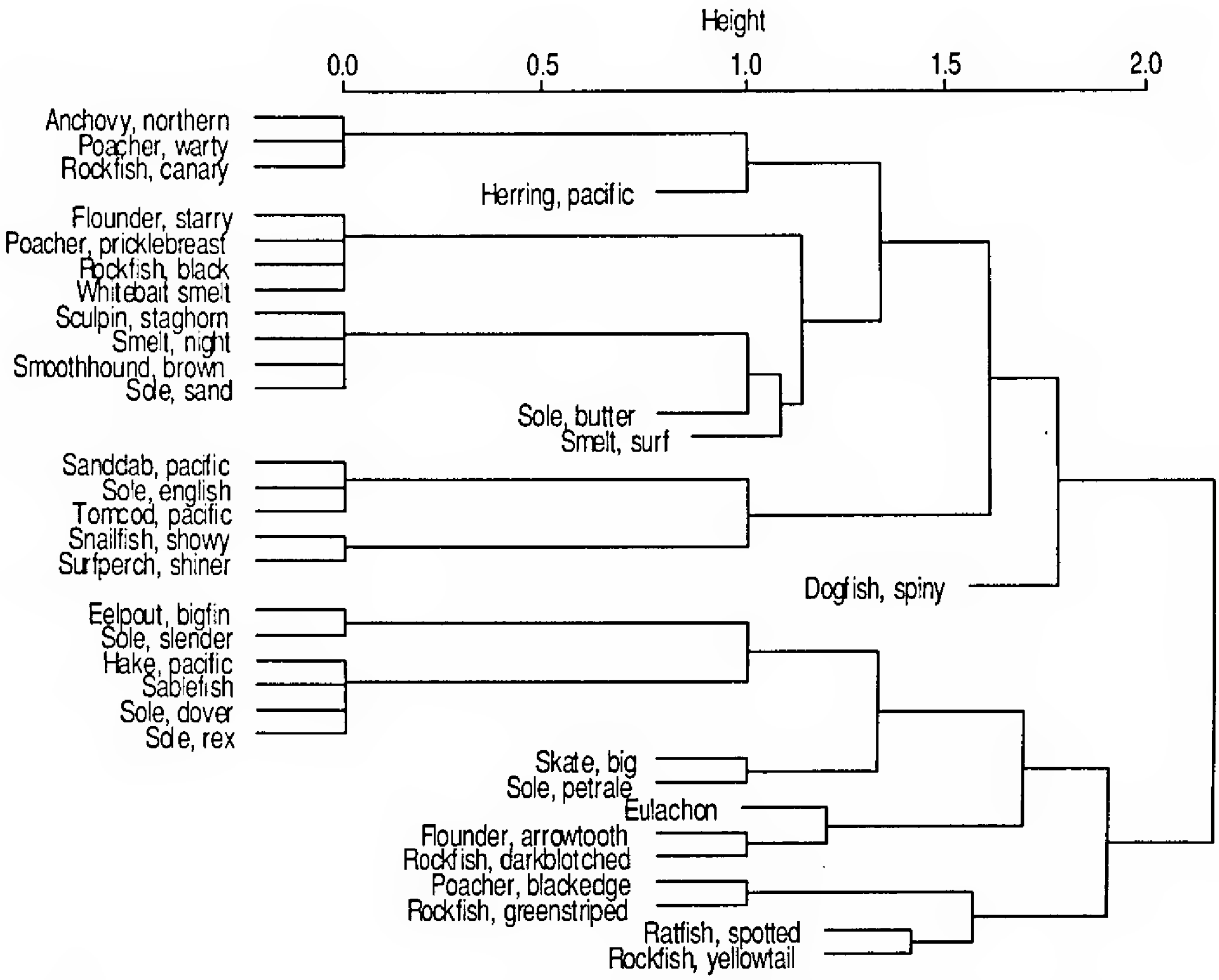

Figure 3. Cluster analysis using frequency of occurrence by depth for depth bins from 25m to 200m by 25m intervals for 1969.

Sport angler landings from 1980 to 1986 included 26 species that we caught in these crab trawls (Karpov et al. 1995). It is reasonable to conclude that most if not all of these fishes caught in trawls, either as juveniles or adults, serve as forage for larger fishes, such as arrowtooth flounder and Pacific hake (Gotshall 1969). There is evidence that populations of several species captured in our trawls have declined. Moyle and Davis (2000) indicate that populations of longfin smelt and eulachon have declined during recent years and are species of "special concern".

Many of the commercially important species captured during the trawling have also declined and have been designated as either overfished (including bocaccio and canary rockfish) or species of concern including black rockfish, widow rockfish, and lingcod (Leet[6]).

Bycatch catch-per-unit-of-effort (CPUE), in some instances, may be a better estimator of population changes than is CPUE obtained from a directed fishery. Fishermen attempt to maximize their catches for a given time or cost by locating areas where targeted species densities are higher. Consequently, CPUE obtained from a commercial fishery may not accurately reflect targeted species abundance. This can be especially troublesome for species that are somewhat residential and which do not

redistribute themselves in response to removals. A number of rockfish may be in this category. Tag returns from most nearshore rockfish show little adult movement over time (Lea et al. 1999). Species exhibiting this behavior may be susceptible to serial depletion showing little reduction in CPUE as harvesters move from heavily fished regions to relatively unfished areas. This happened in the 1980's in the trawl fishery with the development of roller gear that allowed trawlers to fish rocky bottom habitat which they had previously avoided. During this same period, larger vessels fishing midwater trawls entered the fishery. These resulted in short-term CPUEs that were much greater than CPUEs from the fishery prior to the adoption of these trawler gear improvements.

Bycatch, by definition, is the catch of non-targeted species. Consequently, catches will come from marginal as well as primary habitats. When populations increase, they often move into marginal habitat expanding outside of their normal range. Likewise, when populations decline, one might see reductions in fish densities earlier in marginal habitats than in preferred habitats. Such changes may be apparent in bycatch CPUE sooner than in CPUE from fisheries targeting the species.

Cluster analysis results broke out three assemblages that were consistent between 1967 and 1969. These results give a view of species relationships in this region at this time. The value of a single, short-term view of such assemblages is marginal, but comparisons with future surveys may be insightful in following both natural and fishery caused changes in the relative distribution and abundance of assemblage species. There is a confounding effect with trawl tows in that pelagic species may be caught while the net is being deployed and retrieved and consequently are not necessarily strongly associated with the bottom depth being fished.

Recommendations

Commercial shrimp and bottom fish trawler bycatch should be monitored by at-sea observers. We also suggest that all future research surveys identify, count, and obtain length/size data on both targeted and non-targeted species. The extra cost of collecting such bycatch data only marginally increases the total costs of surveys. In situations where large numbers of a species are encountered then statistically valid sampling should be used to estimate total numbers and size distribution of that species in catches. The availability of long term abundance and sizes structure estimates of a population are important in calculating population dynamic parameters and in monitoring status of stocks.

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trawling, but also assisted the biologists in many other ways.

R. N. Lea reviewed the manuscript and pointed out errors in scientific and common names of fishes. A. Gotshall did the word processing.

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AN ASSESSMENT OF THE HAZARD OF A MIXTURE OF THE HERBICIDE RODEO® AND THE NON-IONIC SURFACTANT R-11® TO AQUATIC INVERTEBRATES AND LARVAL AMPHIBIANS

Joel Trumbo
California Department of Fish and Game
1701 Nimbus Road
Rancho Cordova, CA. 95670

This study was conducted to determine whether the aquatic herbicide Rodeo® (active ingredient: glyphosate) and the non-ionic surfactant R-11® (active ingredient nonylphenol polyethoxylate or NPE) adversely affect aquatic species including invertebrates and larval amphibians. A Rodeo®/R-11® mixture was applied directly to the surface of a pond in a manner that would produce atypically high concentrations of these compounds in water. Water samples were collected from the treated pond for chemical analyses and toxicity tests with the aquatic invertebrate *Ceriodaphnia dubia*. A toxicity test with the Rodeo®/R-11® mixture was also conducted to determine the LC_{50} value for the larval life stage of the northern leopard frog, *Rana pipiens*. Water samples collected one hour after application contained the following mean concentrations: glyphosate, 1.83 mg/L; NPE, 1.10 mg/L; and 0.02 mg/L of the NPE breakdown product nonylphenol (NP). Concentrations of glyphosate's primary breakdown product, amino methyl phosphonic acid (AMPA), were below the laboratory detection limit of 0.020 mg/L. Water samples collected from the treated pond were not acutely lethal to *Ceriodaphnia dubia*. The 96-h toxicity test with the Rodeo®/R-11® mixture using *Rana pipiens* produced LC_{50} values of 6.5 mg/L for glyphosate and 1.7 mg/L for NPE, indicating that the mixture is moderately toxic to the amphibian. A comparison of toxic units for the herbicide and surfactant in the mixture indicated that the toxicity to larval frogs was likely due to R-11® and not Rodeo®.

INTRODUCTION

The glyphosate herbicide Rodeo® is commonly mixed with the non-ionic surfactant R-11® to control vegetation growing in or near surface water. R-11® increases herbicide efficacy by improving foliar coverage and by increasing the penetration of the herbicide through the leaf's cuticle layer. Toxicity information for the active ingredient of R-11®, nonylphenol polyethoxylate (NPE), indicates that the compound is moderately toxic to fathead minnows with a 96-h LC_{50} value of approximately 4.5 mg/L (Staples et al. 1998). The NPE metabolite, nonylphenol (NP), is more toxic than its parent compound and, like NPE, exhibits estrogen-like properties.[1] NP has a 96-h LC_{50} value of approximately

[1]Bakke, D. 2003. Human and ecological risk assessment of nonylphenol polyethoxylate-based (NPE) surfactants in Forest Service herbicide applications. USFS internal report. May 2003.

0.13 mg/L, which places the compound in the highly toxic category (Brooke 1993). While the breakdown of NPE generally produces a variety of related compounds with shorter carboxylate chains, NP residues can be produced under low water temperature, nutrient level and dissolved oxygen conditions (Brooke 1993).

The 96-h LC_{50} values for the isopropylamine salt of glyphosate for fish species range from 97 to >1,000 mg/L (Giesy et al. 2000). Based on a U.S. Environmental Protection Agency rating system, the herbicide is considered practically non-toxic to slightly toxic to aquatic species (Zucker 1985). In 1997, the California Department of Fish and Game conducted a study that investigated glyphosate toxicity to larval amphibians (Trumbo 1997). This study demonstrated that larval frogs have a sensitivity to glyphosate that is similar to that of larval fish.

This study assessed the hazard of the Rodeo®/R-11® mixture to two test species. An application of the mixture was made directly to water in order to determine worst-case impacts on non-target aquatic fauna. This study collected information on: 1) The magnitude and persistence of glyphosate, amino methyl phosphonic acid (AMPA), NPE and NP residues in water; 2) the toxicity of water samples collected from the Rodeo®/R-11® mixture application site to the aquatic invertebrate *Ceriodaphnia dubia*; and 3) the 96-h LC_{50} toxicity value of the Rodeo®/R-11® mixture for larval northern leopard frogs, *Rana pipiens*. *Rana pipiens* was selected for this study because of its taxonomic relationship with several other species in the genus *Rana* that have federal or state protected status in California, including the California red-legged frog, *Rana aurora draytonii*.

MATERIALS AND METHODS

Rodeo®/R-11® Mixture Application

The Rodeo®/R-11® mixture was applied to a 0.72 acre-feet pond at the Sacramento-Yolo Mosquito and Vector Control District facility in Elk Grove, California. A similar pond, located nearby, was used as an untreated control. Neither the control or treatment pond had any outflow. The ponds were not connected at the time of the application. Water temperatures during the project ranged from 15° to 18° C. Both ponds had minimal growth of aquatic vegetation and supported populations of mosquitofish, *Gambusia sp.*

The Rodeo®/R-11® mixture application to the pond was made with a vehicle-mounted hose-gun sprayer using typical herbicide and surfactant tankmix concentrations of 1% and 0.5%, respectively. The herbicide use rate was 5 pints/surface acre. In a departure from the standard application technique, the mixture was applied directly to the water surface rather than to the foliage of emersed aquatic weeds. This was done to provide atypically high herbicide and surfactant concentrations in water. Additionally, the lack of outlet flow from the treated pond assured the maximum residency time for chemical residues. All other aspects of the application were made according to the herbicide and surfactant product labels.

Sample Collection

Water samples were collected from three locations in the treated pond and from one location in the non-treated control pond. Each sampling location consisted of a transverse transect with three sampling subsites. The three subsamples collected along each transect were composited into a single sample.

Samples were collected in 500-ml amber glass containers at the water surface by hand or with a polyvinyl chloride (PVC) sampling cup with a handle extension. The sampling cup was cleaned with petroleum ether solvent and deionized water between transects. After collection, samples were immediately stored at a temperature of 4°C, protected from sunlight and then transported directly to the laboratory. One field blank, one matrix spike, and one matrix spike duplicate were collected for quality assurance purposes for every twenty samples collected in the field.

Water samples were collected for chemical analyses and toxicity tests according to the following schedule: one day prior to treatment (Pre-treatment); less than one hour after treatment (Day 0); one day after treatment(1 DAT);two days after treatment (2 DAT); four days after treatment (4 DAT); and eight days after treatment (8 DAT).

Chemical Analyses

Chemical analyses were conducted by the California Department of Fish and Game (DFG) Water Pollution Control Laboratory (WPCL). NPE and NP samples were analyzed using analytical methods developed by Thiele et al. (1997). Glyphosate and AMPA samples were analyzed using high performance liquid chromatography (HPLC) following an accepted analytical method (USEPA 1990).

Acute Toxicity Tests

Toxicity tests were conducted by the DFG Aquatic Toxicology Laboratory (ATL). All laboratory procedures were done in compliance with Good Laboratory Practices, in strict observance of the ATL Quality Assurance Manual.

An acute toxicity test was conducted to determine the 96-h LC_{50} value for the Rodeo®/R-11® mixture using larval northern leopard frogs, *Rana pipiens*. Test frogs were purchased from Carolina Biological Supply and were < 7-d old during the test. Four replicates per treatment and a negative control (laboratory well-water) were used. The larvae were exposed in 1000-ml beakers, each containing 250 ml of test solution. The test was conducted at a temperature of 22°C, and there were 10 larvae per replicate (40 larvae per treatment). The tests were conducted in environmental chambers under constant temperature, humidity and photoperiod conditions. Test solutions were renewed at 48 hours. Samples of test solutions were collected and analyzed at the beginning of the test to verify the Rodeo® and R-11® exposure levels.

Water samples were collected from one location in the treated pond and from one location in the non-treated control pond prior to the application and on Day 0 and 4 DAT

to test for toxicity to *Ceriodaphnia dubia*. Standard 96-h toxicity tests with 48-h renewals were completed using accepted methods (USEPA 1993).

RESULTS

Environmental Exposure Levels

Rodeo®

Analysis of water samples collected prior to the herbicide/surfactant application did not reveal any detectable glyphosate residues in the test ponds. One hour after the Rodeo®/R-11® mixture application, the mean glyphosate concentration in the treated pond was 1.83 mg/L (Table 1), with a maximum concentration of 3.1 mg/L. Glyphosate concentrations in the treated pond declined rapidly during the first 24 hours after the application. At 1 DAT, mean glyphosate residues had declined by more than 84%. Glyphosate residues in samples collected 4 and 8 DAT remained stable at 0.2 mg/L. AMPA residues were never detected in the treated pond above the laboratory minimum detection limit of 0.02 mg/L. The untreated control pond contained no detectable residues of either glyphosate or AMPA.

R-11®

Within one hour after the Rodeo®/R-11® mixture application, the mean NPE and NP concentrations in the treated pond were 1.1 and 0.020 mg/L respectively (Table 1). The maximum concentrations of these compounds were 1.8 and 0.03 mg/L, respectively. Chemical analyses prior to the application indicated the water in both the treatment and control ponds contained low concentrations of NPE and NP. The minimum detection limit (MDL) for both compounds was 0.0002 mg/L.

NPE concentrations in the treated pond declined rapidly from those detected one hour after application. After 24 hours, NPE concentrations had been reduced by 64%, and by 4 DAT the peak concentration had been reduced by more than 98%. By 8 DAT the NPE concentration in the pond had been reduced to nearly the pretreatment level.

Residues of the NPE metabolite, NP, were markedly lower than those of its parent compound one hour after the Rodeo®/R-11® mixture application (0.020 mg/L vs 1.1 mg/L). By 1 DAT, NP residues had been reduced by 75%, and by 4 DAT they had been reduced by 95%. By 8 DAT, the NP concentration had been reduced to nearly the pretreatment level.

Table 1. Mean concentrations of glyphosate, AMPA, NPE and NP, after application of a Rodeo®/R-11® mixture.

	Glyphosate(mg/L)	AMPA(mg/L)	NPE(mg/L)	NP(mg/L)
Pretreatment	ND[a]	ND	0.003	0.001
1-H POST	1.830	ND	1.100	0.020
1 DAT[b]	0.300	ND	0.400	0.005
2 DAT	0.300	ND	0.040	0.002
4 DAT	0.200	ND	0.020	0.001
8 DAT	0.200	ND	0.004	0.001

[a]ND indicates no detectable residues $\geq$ the minimum detection limit (MDL). MDL values are as follows: NPE, 0.0002 mg/L; NP, 0.0002 mg/L; glyphosate, 0.02 mg/L; and AMPA, 0.02 mg/L;
[b]DAT = Days after treatment

Acute Toxicity Tests

Rodeo®/R-11® Mixture - The calculated 96-h LC_{50} values for larval northern leopard frogs were 6.5 mg/L glyphosate and 1.7 mg/L NPE (Table 2).

Table 2. Calculated LC_{50} values and mean characteristics of chemical dilutions and survival of larval *Rana pipiens* in 96-h acute toxicity test.

Dilution No.	Glyphosate(mg/L)	NPE(mg/L)	NP(mg/L)	Survival(%)
1	17.6	4.5	<MDL	0*
2	8.7	2.5	<MDL	0*
3	4.5	1.0	<MDL	92.5
4	2.4	0.6	<MDL	100
5	1.3	0.3	<MDL	100
96-h LC_{50}	6.5	1.7	NA	NA

* Indicates survival significantly less than control group ($P<0.05$).

Treated Pond Water - There was no significant mortality of *Ceriodaphnia dubia* in any of the water samples collected from the treated pond. Cladoceran survival ranged from 90 to 100% in the pond water samples and from 95 to 100% in the laboratory controls.

DISCUSSION

Glyphosate, NPE and NP Environmental Exposure Levels

Initial concentrations of glyphosate and NPE were relatively high, particularly when compared to residue levels that result from more typical application scenarios (i.e., applications made to emersed aquatic plants and not directly to water). For example, monitoring conducted by the California Department of Boating and Waterways (DBW) as part of their water hyacinth control program generally failed to detect glyphosate and NPE residues in samples collected within 24 hours after Rodeo®/R-11® applications (California Department of Boating and Waterways 2003).

It is difficult to assess the relative magnitude of the NPE and NP residues that were observed during this study because little historical data are available on the residues of these compounds after they have been applied directly to water. However, the glyphosate residues that were observed are consistent with those seen in a previous study conducted by Horner[2]. In that study, 1.7 mg/L glyphosate was detected in pond water after an herbicide application.

The persistence of glyphosate, NPE and NP was in general agreement with what had been demonstrated by previous investigations. Research by Trumbo (2002) showed that peak concentrations of the three compounds in water were achieved relatively soon after application and declined rapidly during the first 24 hours. While glyphosate residues were still detectable in the pond eight days after treatment, this is not inconsistent with what is known about the herbicide. Environmental fate information for glyphosate indicates that the compound's half-life in water can range from 12 days to 10 weeks (USEPA 1992). Residues of NPE and NP were more transitory than the herbicide residues with concentrations for both compounds approaching pretreatment levels by eight days after the herbicide/surfactant application. While the source of the pretreatment detections of NPE and NP is unknown, it may be related to previous uses of surfactant products for terrestrial weed control near the ponds.

Aquatic Toxicity Hazard

The hazard of the Rodeo®/R-11® mixture to aquatic life is largely determined by the concentration of R-11® (NPE and NP) because it is the more toxic compound in the tankmix. Although glyphosate can be toxic at levels in excess of 500 mg/L, R-11® can be toxic at approximately 1 to 6 mg/L (Table 4). When the two products are tested together in a 2:1 mixture, the toxicity of R-11® changes little (mean LC_{50} value decreases by 1.8x), but the toxicity of glyphosate changes dramatically (mean LC_{50} value decreases by 208x). When toxic units (LC_{50} mixture/LC_{50} individual chemical) for each compound are calculated, the values for NPE are greater than glyphosate's.

It is not surprising that there was no significant mortality to cladocerans in the water

[2]Horner LM. 1990. Dissipation of glyphosate and aminomethyl phosphonic acid in forestry sites. Unpublished report MSL-9940. Monsanto Company, St Louis, MO.

samples collected from the treated pond. The concentrations of NPE and NP one hour after application were about 50% of their LC_{50} values for this species, and concentrations of both compounds decreased significantly by one day after the herbicide/surfactant application.

Table 4. Comparison of LC_{50} values(mg/L) and toxic units for various invertebrate and vertebrate species to Rodeo®, R-11® and a Rodeo®/R-11® mixture.

Species	Rodeo®[a]	R-11®[b]	Rodeo®/R-11® Mixture	Toxic Units[c]
Ceriodaphnia dubia[d]	586	5.7	3.1/2.8	Rodeo: <0.01 R-11: 0.49
Fathead minnow[e]	652	1.1	2.8/0.9	Rodeo: <0.01 R-11: 0.82
Sacramento splittail[f]	1132	3.9	5.5/2.1	Rodeo: <0.01 R-11: 0.54
Leopard frog[g]	—	—	6.5/1.7	—

[a] As glyphosate.
[b] As sum of NPE and NP.
[c] Toxic Units = LC_{50} mixture/LC_{50} individual chemical.
[d] California Department of Fish and Game, Aquatic Toxicology Laboratory Report P-2376, Elk Grove California.
[e] California Department of Fish and Game, Aquatic Toxicology Laboratory Report P-2365, Elk Grove California.
[f] California Department of Fish and Game, Aquatic Toxicology Laboratory Report P-2369, Elk Grove California.
[g] From this study.

CONCLUSIONS

The results of this study demonstrate that the application of a Rodeo®/R-11® mixture directly to water at labeled rates can produce relatively high glyphosate, NPE and NP residues that decline rapidly within 24 hours after application. While undiluted Day 0 samples collected from the treated pond did not produce significant mortality of the aquatic invertebrate *Ceriodaphnia dubia* in laboratory tests, the initial concentration of NPE in the pond was approximately equal to the 96-h LC_{50} value for larval northern leopard frogs that was determined during this study. Glyphosate and NP residues, even at their highest concentrations immediately after application, never exceeded 29% and 15%, respectively, of their established acute toxicity values for larval frogs.

Future areas of research could include testing the toxicity of recently treated water to larval frogs. Also, increasing the frequency of sample collection on Day 0 would be useful for determining the persistence of the chemicals. This information would also

prove valuable in establishing if NPE toxicity to non-target aquatic species occurs primarily during the first 24 hour period as other researchers have suggested.

Finally, additional monitoring studies for the NPE metabolite NP would also be important, particularly in light of the compound's relatively high toxicity and its identification by the U.S. Department of Agriculture as a potential endocrine disruptor.[3]

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[3]Bakke, D. 2003. Human and ecological risk assessment of nonylphenol polyethoxylate-based (NPE) surfactants in Forest Service herbicide applications. USFS internal report. May 2003.

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THE EFFECT OF DIETARY SUPPLEMENTED L-ARGININE ON THE GROWTH OF JUVENILE HATCHERY REARED WHITE SEABASS, *ATRACTOSCION NOBILIS*

ERIC F. MILLER
Department of Biology
California State University, Northridge
18111 Nordhoff Street
Northridge, CA 91330

MICHAEL P. FRANKLIN
Department of Biology
California State University, Bakersfield, Antelope Valley Branch Campus
43909 30th Street West
Lancaster, CA 93536-52092

INTRODUCTION

White seabass, *Atractoscion nobilis*, is the largest of the member of the family Sciaenidae commonly occurring off the California coast (Miller and Lea 1974). White seabass are prized by both the commercial and recreational fishing communities within southern California (Vojkovich and Crooke 2001). Fishing pressure, both commercial and recreational, has severely impacted the local stocks. Total landings (commercial and recreational) have steadily declined since the 1950's despite several management strategies (e.g., minimum size requirements, catch limits, etc.) (Vojkovich and Crooke 2001). In the late 1990's, the annual recreational landings had fallen below 5000 fish, a significant drop in comparison to the estimated 33,000 landed fish in 1947 (Vojkovich and Crooke 2001).

Efforts to alleviate the demise of coastal stocks of white seabass were undertaken by the California Department of Fish and Game with the initiation of the Ocean Resource Enhancement Hatchery Program (OREHP) in 1983. The goal of this program was to develop aquaculture techniques in order to produce juvenile white seabass to be released into coastal California waters in order to directly supplement the natural stocks (Vojkovich and Crooke 2001). By 1999, over 375,000 juvenile white seabass were released into the wild (Vojkovich and Crooke 2001).

The non-essential amino acid L-arginine and its stimulatory effects on growth hormone release have received considerable study in mammals (Fajans and Floyd 1972; Swanson 1990). Hird (1986) notes ubiquitous use of L-arginine in metabolic pathways: protein synthesis, urea production, interrelation with the metabolism of glutamic acid and proline, synthesis of creatine and polyamines. The process of urea production in marine fish was found to be greatly reduced, thereby possibly rendering L-arginine a greater role in protein synthesis than is found in terrestrial animals (Hird 1986).

The effect of this amino acid on olfactory stimulation has also received considerable attention. Caprio (1988) conducted electrophysiological studies on olfactory receptors and found a high sensitivity to several amino acids in many fish species. Carr et al. (1977) found amino acids to be strong stimuli of feeding responses in the course of behavioral studies of fish feeding. Ishida and Kobayashi (1992) identified L-arginine as a potent stimulant to olfactory receptors in rabbitfish, *Signaus fuscescens*.

The beneficial effects of L-arginine on the growth of juvenile fish growth has been documented for several species (Lopez-Alvarado and Kanazawa 1994, Lall et al. 1994, Moon and Gatlin 1994, Tibaldi et al. 1994, Murillo-Guerra et al. 2001). Specific attention has been given to commercially important cultured species, such as Atlantic salmon, *Salmo salar*, European sea bass, *Dicentrarchus labrax*, and red drum, *Sciaenops ocellatus* (Lall et al. 1994, Moon and Gatlin 1994, Tibaldi et al. 1994).

Results of previous studies on captive populations showed that diets either naturally high, or supplemented with L-arginine elicited higher growth rates than diets containing lower quantities of the amino acid. Direct dietary supplementation enhanced growth in a pattern corresponding to the L-arginine level until a species-specific threshold was reached (Lopez-Alvarado and Kanazawa 1994, Lall et al. 1994, Tibaldi et al. 1994). Moon and Gatlin (1994) fed red drum, also of the family Scaenidae, natural diets available to the carnivorous fish in the wild. This study found the diet highest in L-arginine content yielded an overall growth rate of 713% over the period of observation, the highest of all diets tested.

To determine the effectiveness of L-arginine in enhancing the growth of juvenile hatchery reared white seabass, we fed two populations diets supplemented with varying levels of L-arginine. The mean weight of each population was monitored as a measure of growth (Baker 1986).

MATERIALS AND METHODS

Fifty white sea bass were held in two separate tanks (25 individuals per tank) over a period of 75 days. The individuals in each were fed two diets based on the standard fish-meal pellet produced by Moore-Clark™[1] used by the Hubbs-Sea World Research Institute (HSWRI) hatchery in Carlsbad, California. Pure L-arginine was added to the pellets at a ratio of 3.50% of total food weight for the treatment tank. The addition of L-arginine was done in a process similar to that used by Lopez-Alvarado and Kanazawa (1994). The L-arginine was added to powdered fish-meal, coated with liquid coconut oil and allowed to cool. Coconut oil seals the L-arginine to the fish-meal to prevent leaching off, but posed no toxicity problems (S. Folwkes[2] *CERI* personal communication). The 0.00% L-arginine control diet was also coated in the same fashion. Both diets were then formed into small (<10 mm^3) pellets and dispersed into the proper tanks at 4 g per

[1]Does not constitute endorsement by California Department of Fish and Game.
[2]Steven Folwkes, Cognitive Enhancement Research Institute, San Francisco, CA, April, 1997

individual fish per tank. Feedings occurred twice daily, morning and evening, during the course of the study.

Both populations were maintained in separate flow-through tanks supplied from a unique water source. Water quality was monitored in both tanks for the duration of the study: Dissolved oxygen = 7.4 ppm, pH = 8.00, salinity = 34 ppt, and temperature = 17 °C - 22 °C (range corresponded to air temperature). Ten individuals were selected at random biweekly from each population and weighed live. Each fish was weighed individually in a tared 500 mL full beaker of seawater on a digital scale to the nearest 0.01 g.

All statistics were analyzed via SYSTAT v. 9.0. Weights were log transformed prior to statistical analysis due to high heteroscedasticity based on high skewness and kurtosis (Sokal and Rohlf 1995). Weights at day 0 and day 75 were analyzed via two sample t-tests. Total growth over the sampling period was analyzed with a one-way analysis of variance (Sokal and Rohlf 1995).

RESULTS

Both the control and L-arginine treatment populations had an initial mean weight of 1.00 g (t = -0.147, df = 18, p = 0.885) (Table 1). The mean weight of the treatment tank reached 3.31 g by day 75, more than doubling the control tank mean weight of 1.62 g at day 75 (t = 2.422, df = 18, p = 0.028) (Table 1). By day 25 the L-arginine treatment tank exhibited positive growth while the control group exhibited a slight decrease in mean weight. The control group began to show positive growth by day 40, while the L-arginine treatment mean weight decreased slightly at day 40, then exhibited steady positive growth for the remainder of the study period. The addition of L-arginine significantly increased the growth rate of white seabass over the course of the study (F = 12.392, df = 98, p = 0.001) (Figure 1).

Table 1. Growth of white seabass juveniles by L-arginine treatment.

	Percent added L-arginine	Initial Mean Weight (g)	Final Mean Weight (g)
Control	0.00%	1.00	1.62
Treatment	3.50%	1.00	3.31*

*Significance at the p = 0.028 level

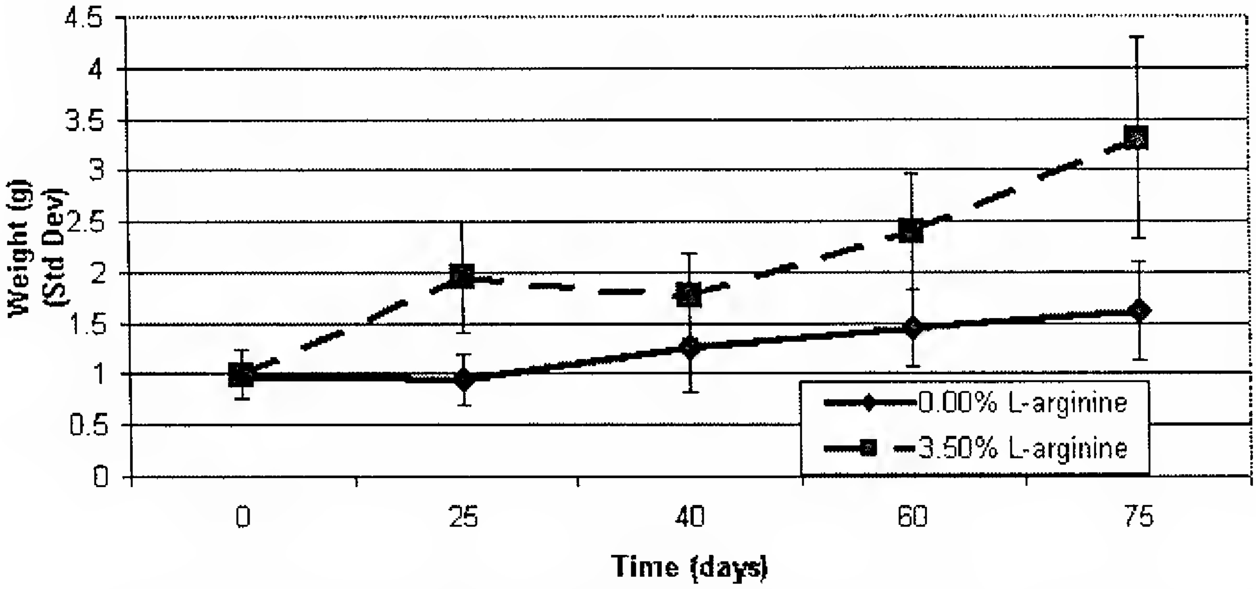

Figure 1. Mean weight of white seabass by L-arginine treatment over time. Vertical bars indicate ± 1 standard deviation.

DISCUSSION

A significantly higher growth rate in juvenile white seabass was stimulated with dietary supplementation of 3.50 % L-arginine based on total food weight. The increase was over double the growth rate shown by the control group (0.00% L-arginine) during the period tested. This data suggests that L-arginine provides measurable effects on the growth of juvenile white seabass. The exact mechanism responsible for the growth was not measured in the course of this study. It has been previously documented that L-arginine positively effects growth hormone release (Fajans and Floyd, 1972; Swanson, 1990) as well as eliciting a strong olfactory response (Isihida and Kobayashi, 1992). An increase in serum growth hormone would result in a significant increase in somatic growth. The same is true of a heightened feeding response triggered by an olfactory stimulation. The data suggests a diet reformulated to include 3.50% L-arginine by food weight would significantly enhance the growth of hatchery reared white seabass.

Growth of marine fish has been previously enhanced by the addition of elevated L-arginine level to the normale diet of cultured fish. Lopez-Alvarado and Kanazawa (1994) fed larval red sea bream diets enriched with L-arginine in levels of 2.3%-3.1%. They found that growth rates increased correspondingly with L-arginine content. Tibaldi et al. (1994) studied the effects of varying dietary L-arginine levels on growth of European sea bass and found an increase in growth rate that was proportional to dietary L-arginine content. Lall et al. (1994) tested the L-arginine dietary requirements of Atlantic salmon diets, and found that supplementation at 5.4% L-arginine per total protein gave the highest growth rate.

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MODIFICATIONS TO AN AGRICULTURAL WATER DIVERSION TO PERMIT FISH ENTRAINMENT SAMPLING

ZOLTAN MATICA and MATT NOBRIGA
California Department of Water Resources
Division of Environmental Services
3251 S Street
Sacramento, CA 95816-7017
zoltan@water.ca.gov

Water diversions in California's Sacramento – San Joaquin Delta can pose a substantial hazard to aquatic fauna (Hallock and Van Woert 1959, Arthur et al. 1996). The largest water diversions in the Delta are the federal facility at Tracy, the state facility at Clifton Court and the Mirant Corporation power plants in Antioch and Pittsburg. Cumulatively, these facilities can divert more than 60% of the total Delta inflow and are equipped with fish barriers of varying efficiency (Brown et al. 1996). Additionally, approximately 2,200 smaller agricultural diversions remove an average of about 27% of June-July inflows (White and Kawasaki[1] 1998). Many of these agricultural diversions are located within the boundaries of critical habitat for listed species (DFG[2] 2000) such as delta smelt, *Hypomesus transpacificus* (USFWS[3] 1993), steelhead, *Oncorhynchus mykiss*, and winter-run Chinook salmon, *O. tshawytscha* (NMFS[4] 2000). Most of these smaller diversions are unscreened siphons, 20 to 46 cm in diameter, that draw water 60 to 90 cm above the channel bottom (Herren and Kawasaki 2001).

Unfortunately, little data is available on the effects of small diversions on fish in the Delta. A major limitation is that most conventional sampling gear is designed to work in open water areas and does not work effectively at high velocity agricultural diversions. Another problem is that it is hard to evaluate the potential benefits of adding fish screens; because there is only limited data on fish entrainment rates through unscreened diversions, it is unclear how many fish might have been saved if a screen had been in place.

Here, we report on structural modifications at the outlets of a California Department of Water Resources' (DWR) agricultural diversion facility. The facility is located on

[1] White, J.R. and S.S. Kawasaki. 1998. Inventory of water diversions in four geographic areas in California. (Draft) California Department of Fish and Game, Sacramento, California.

[2] DFG (California Department of Fish and Game). 2000 April. State and federally listed endangered and threatened animals of California. April 2000 [web page]. Available from: http://www.dfg.ca.gov/endangered/t_e_animal.pdf, via the INTERNET. Accessed 2000 December.

[3] USFWS (U. S. Fish and Wildlife Service). 1993. Final rule listing the delta smelt as a threatened species. Federal Register 5 March 1993 (58 FR 12854).

[4] NMFS (National Marine Fisheries Service). 2000. Designated critical habitat: Critical habitat for 19 evolutionarily significant units of salmon and steelhead in Washington, Oregon, Idaho, and California. Federal Register 16 February 2000 (50 CFR 226).

the lower Sacramento River at Horseshoe Bend, an area of typically high delta smelt density (Sweetnam 1999). Using siphons, the facility diverts river water over the levee for irrigation typically from April through July. To comply with requirements of the U.S. Army Corps 404 Permit and the U.S. Fish and Wildlife Service's Biological Opinion on delta smelt, the facility was rebuilt in 1997 with two screened pipes, an unscreened pipe, and structural modifications to facilitate studies of delta smelt entrainment. The modified facility allows efficient sampling where, (1) 100% of the diverted water can be filtered by nets, and (2) adjacent screened and unscreened pipes can be sampled, allowing accurate evaluation of the benefits of screening (Nobriga et al. 2003).

The facility consists of an intake structure on the river side and an outlet structure on the inland side at the base of the levee. The intake structure supports a screen backwash pump, flow control valves, two screened 61-cm pipes and one unscreened 61-cm pipe that is modified to accept a future fish screen. This unscreened siphon is in-line with, but 2.3 m downstream from, the nearest fish screen. The unscreened siphon is only used during fish entrainment sampling (Nobriga et al. 2003) or when there are mechanical problems with the screened siphons. The screened pipes are joined at an intake manifold equipped with two cylindrical fish screens manufactured by Custom Technologies Company[5], Inc. The screens are 1.5 m long, have a radius of 1.0 m, and a mesh size of 2.4 mm. The centerline of the intake is 1.5 m below the mean low water mark and the screens are 0.6 – 0.9 m off the bottom. Maximum approach velocity at the screens is 6 cm/s (~0.2 ft/s) when both screened pipes are operated at their theoretical maximum of $0.42\ m^3/(s\cdot pipe)$.

The following describes the modifications done to the Horseshoe Bend facility outlet structure in order to support fish entrainment sampling. Access over the 0.2-ha outlet pool to the screened and unscreened siphon outlets was provided by a catwalk and a sampling platform (Figure 1). Nets can be attached to coupling necks on each of the two vertical sliding gates. One sliding gate positions a net over the unscreened outlet and the other positions a second net over the screened outlet. The gates are raised or lowered into place by hand winches. The volume of water sampled can be estimated using propeller flow meters suspended by steel studs in the center of each sliding gate neck.

The custom-designed plankton nets (Figure 2) have mouths modified with canvas collars to encompass 3-point coupling rings. The rings connect with pins to the 61-cm diameter necks protruding from the sliding gates. The 1600-µm mesh plankton nets are 5.2 m long, 1.8 m in greatest diameter and have removable PVC cod-ends. Each net has two aluminum spreader hoops, one 1.8-m diameter hoop 2.1 m from the mouth and one 1.1-m diameter hoop 3.4 m from the mouth. The cod-end is a removable PVC collection tube 0.4 m long with a 0.2 m diameter and has 1600-µm mesh-covered openings on its sides. Each hoop and cod-end has bullet floats attached to properly align the nets during sampling. A tender line is attached to the tops of each hoop and the cod-end to facilitate retrieval.

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[5] Use of trade names does not imply endorsement by the California Department of Water Resources or by the California Department of Fish and Game.

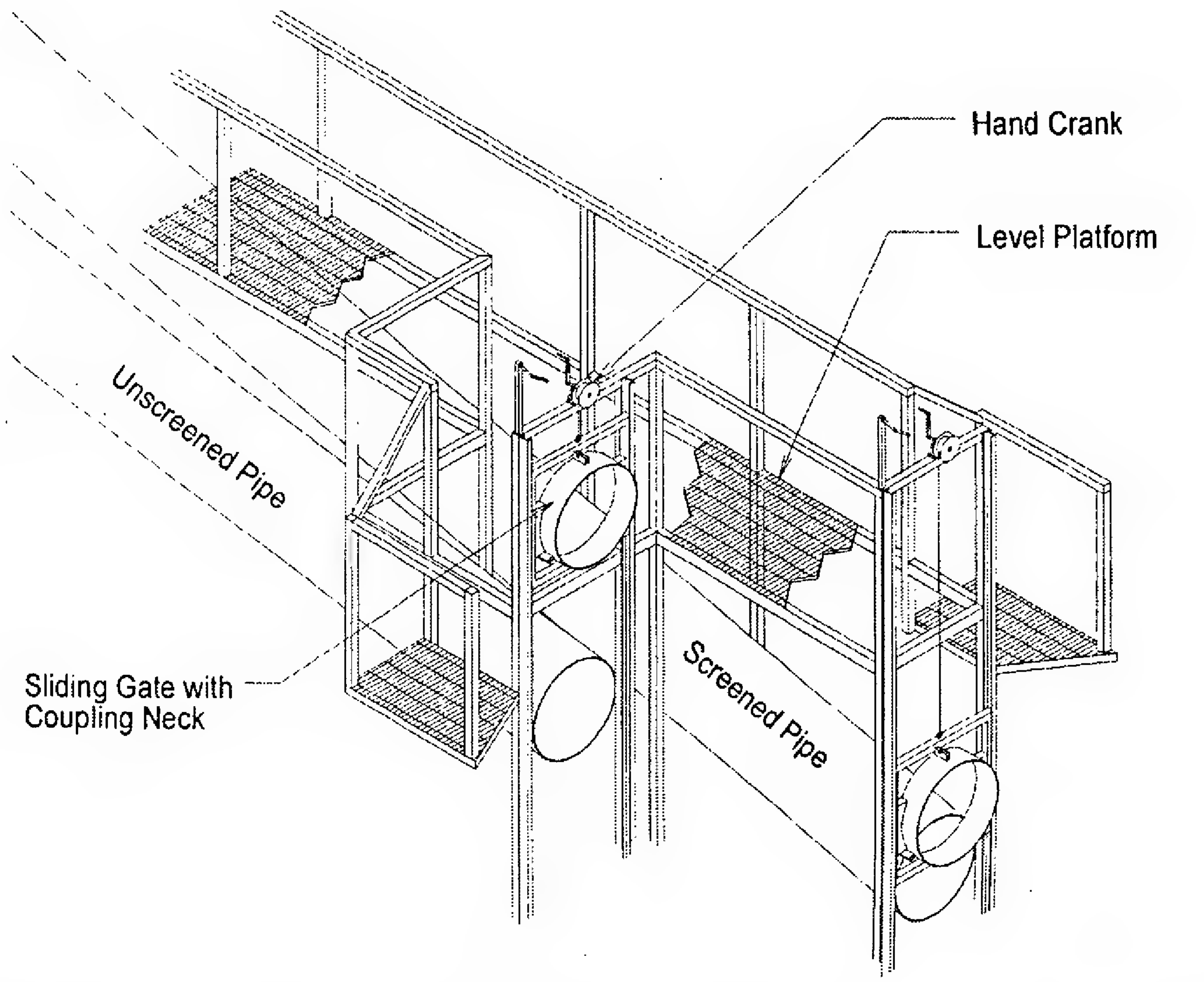

Figure 1. Outlet sampling structure with net coupling sliding gates for a screened and unscreened pipe (R. Beckwith).

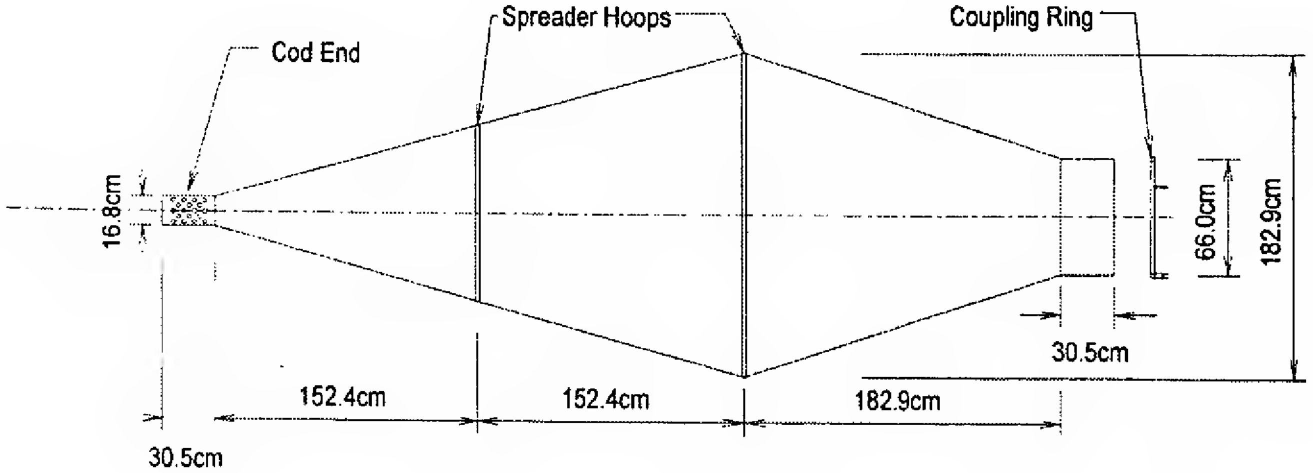

Figure 2. Modified fyke net with two spreader hoops and a coupling ring (R. Beckwith).

The outlet sampling structure has been in place for 4 years with no operational problems. Fish entrainment sampling conducted by DWR during the summer of 2000 and 2001 demonstrated the effectiveness of the modifications to the facility (Nobriga et al. 2003). It should be noted that the Horseshoe Bend site is somewhat unusual among Delta agricultural diversion facilities, in that it is owned by DWR. In contrast, most facilities are privately owned and operated. Therefore, construction of similar facilities is likely to require cooperation of landowners or leaseholders.

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RECORDS OF CHAMELEON GOBY, *TRIDENTIGER TRIGONOCEPHALUS*, IN SAN DIEGO BAY, CALIFORNIA

Daniel J. Pondella, II
Vantuna Research Group
Moore Laboratory of Zoology
Occidental College
1600 Campus Road
Los Angeles, CA 90041
Pondella@oxy.edu

Zachary K.J. Chinn
Department of Biology
Occidental College
1600 Campus Road
Los Angeles, CA 90041

Tridentiger trigonocephalus, the chameleon goby, is native to the Asian Pacific and was inadvertently introduced into California from the Orient (Miller and Lea 1976). Evidence suggested that ballast water by international ships was the cause (Matern and Fleming 1995), however, Dill and Cordone (1997) also suggested that eggs in Pacific or Japanese oysters could be the basis for introduction into San Francisco Bay. Males brood demersal eggs (Watson 1996), which have an incubation time of 8.5 days (Haaker 1979). Haaker (1979) also proposed that these eggs on fouling organisms might be a transpacific transport mechanism. Chameleon goby do not grow large; seldom-surpassing 90 mm TL, they are found in shallow bay areas, and have been noted to live in discarded bottles (Haaker 1979). The first observation of a chameleon goby from California waters was in 1960 in Los Angeles Harbor and chameleon gobies were collected in Los Angeles Harbor along 'rip-rap' rocky-reefs in 1977 (Haaker 1979). Chameleon gobies were first recorded in San Francisco Bay in 1962 (Brittan et al. 1963) and remained common there (Matern and Fleming 1995, Moyle 2002). Considering their disjunct distribution and preference for estuarine or harbor habitats, it appears that chameleon goby has been introduced to the California coast on several occasions.

Here we report on three chameleon gobies that were collected in the San Diego Bay: one in January 1995 and two on 6 January 1998 (Figure 1). The first recorded goby (SIO 00-5) in San Diego Bay was caught during an intensive five-year study of ichthyofauna in this bay, which focused on soft bottom and eelgrass habitats (Allen et al. 2002). It was the only chameleon goby collected in this survey and was caught in an eelgrass station proximal to the Coronado Keys in upper San Diego Bay. Water temperature was 15° C and a salinity 34 ‰ (lat. 32° 39.083' N, long. 117° 08.500' W). Daniel J. Pondella, II collected two chameleon gobies (SIO 98-262) on 6 January 1998. These fishes were collected on a riprap rocky reef, which was created as a fishery enhancement structure proximate to North Island in outer San Diego Bay (lat. 32° 42.254' N, long. 117° 13.471' W). The water temperature and salinity were 15.8° C and 36.8 ‰, respectively. Counts

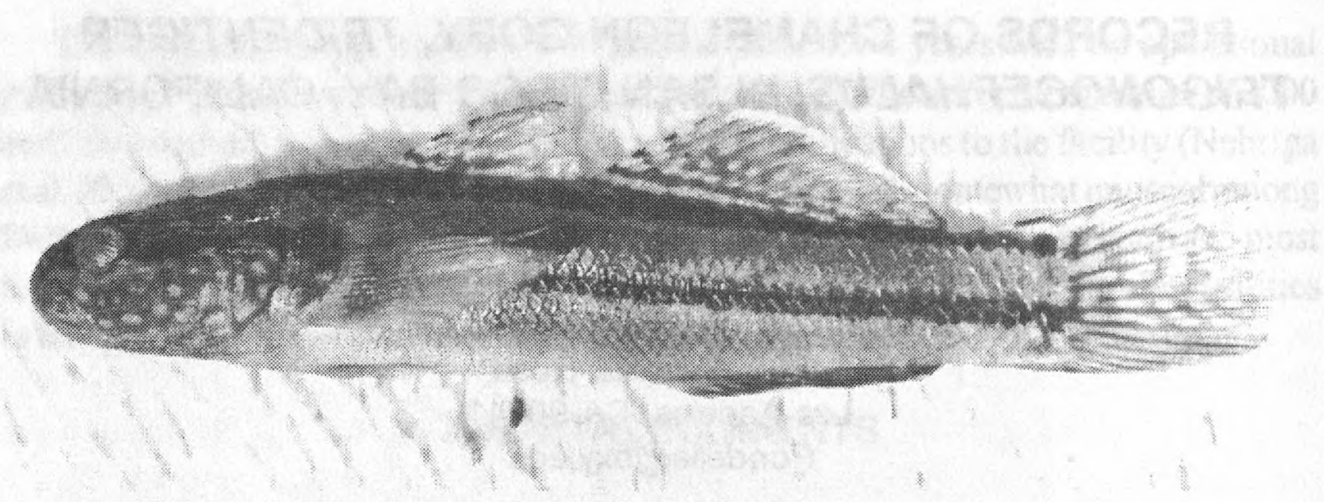

Figure 1. Left lateral view of a chameleon goby, *Tridentiger trigonocephalus*, (48 mm SL, SIO 98-262) collected in San Diego Bay on January 6, 1998. Photograph by Daniel J. Pondella, II.

and characters are given in Table 1 in accord with previous descriptions (Miller and Lea 1976, Matern and Fleming 1995).

It appears that there was a low abundance of chameleon gobies during this period (1994-1999) in San Diego Bay (Allen et al. 2002). However, the major focus of this survey was soft bottom and eelgrass habitats. This could suggest that chameleon goby if present were either predominantly in rocky-reef habitats and picked up incidentally during this survey or simply not abundant in San Diego Bay. These collections suggested either the expansion of the species from Los Angeles Harbor or perhaps they were a result of shipping through one of the various mechanisms previously described.

Table 1. Characteristics of three chameleon gobies, *Tridentiger trigonocephalus*, captured in San Diego Bay.

	Specimens		
	SIO 00-5	SIO 98-262	SIO 98-262
Collection Date	January 1995	6 January 1998	6 January 1998
Standard length (mm)	63	48	43
Total length	76	58	52
Dorsal fin spines, rays	VI + I, 11	VI + I, 12	VI + I, 12
Anal spines, rays	I, 11	I, 10	I, 12
Predorsal scales	21	19	19
Midlateral scales	55	52	50
Upper ray of pectoral fin	Attached	Attached	Attached
Size of sensory canal pores (large or small)	Large	Large	Large
Color of head speckles	White	White	White
Head speckles extend to ventral surface	No	No	No
Dorsal and anal fin edge color	White	White	White

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